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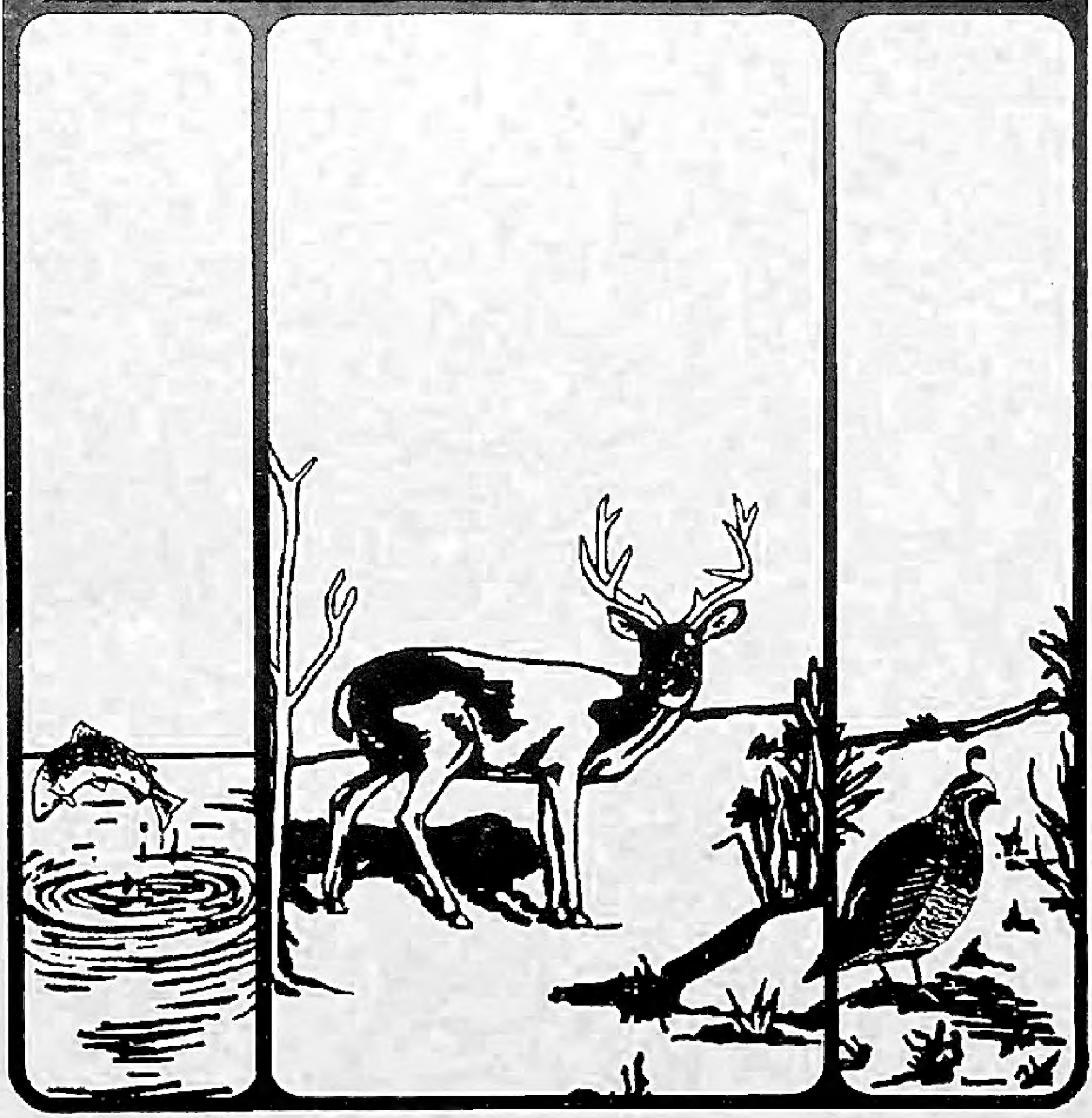

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ARTICLES

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ADULT RECOVERIES OF WINTER AND SUMMER STEELHEAD BY RELEASE LOCATION ON THE LEWIS RIVER, WASHINGTON

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Because of concern for negative interactions between hatchery steelhead, *Oncorhynchus mykiss*, and wild fall chinook, *O. tshawytscha*, in the Lewis River, Washington, a 1988 agreement between Pacific Power, Washington Department of Fisheries (WDF) and Washington Department of Wildlife (WDW) (now combined Washington Department of Fish and Wildlife) required a study whereby steelhead smolts were released at downstream locations for 2 years. Pacific Power was obligated to build and operate a mitigation hatchery for steelhead and sea-run cutthroat trout, *O. clarki*, due to the dams on the river. Staff from WDF was concerned that steelhead smolts released from the chosen upriver hatchery site might prey on wild fall chinook salmon fry. McIsaac[1] (1990) reported that the main juvenile chinook salmon rearing area on the Lewis River was between rkm 11 and 25. Releasing steelhead smolts downstream of rkm 11 was thought to reduce their predation opportunity.

However, staff from WDW was concerned that downstream releases of steelhead smolts might impair the ability of the hatcheries to collect brood stock. Hatchery steelhead return location has been shown to be influenced by smolt release location (Wagner 1969, Cramer 1981, Leider 1989, Slaney et al. 1993, Kenaston et al. 2001). In two rivers about twice as long as the Lewis River, Wagner (1969) found that winter steelhead upper-river releases resulted in more upper-river adult returns and lower-river releases resulted in more lower-river adult recoveries. On a river of similar length as the Lewis River that also had an upriver hatchery, Slaney et al. (1993) found small but significant effects of release location on distribution of adult winter steelhead recoveries but also observed substantial dispersal of lower-river-released fish to the upper river. On a stream about three times the length of the Lewis River, Cramer (1981) found that adult summer steelhead moved to their respective smolt liberation sites and remained in those areas throughout the summer. Kenaston et al. (2001) found that prerelease acclimation and direct smolt releases similarly affected adult winter steelhead distributions.

[1] McIsaac, D.O. 1990. Factors affecting the abundance of 1977-1979 brood wild fall chinook salmon in the Lewis River, Washington. Ph.D. Dissertation, University of Washington, Seattle, Washington, USA.

After the above study was initiated, Hawkins and Tipping (1999) found that hatchery steelhead smolts contained 0.01 to 1.13 salmonid fry per stomach on the Lewis River. Although Hawkins and Tipping (1999) did not estimate the number of wild chinook fry consumed by hatchery steelhead smolts, fry consumption was a concern due to the large number (250,000) of smolts released. The hatchery program of sea-run cutthroat trout on the Lewis River was terminated after the 1999 release, due in part to their frequent consumption of wild chinook salmon fry. Lewis River wild winter steelhead and wild fall chinook salmon were listed as threatened under the Endangered Species Act in March 1998 and March 1999, respectively; hatchery stocks of summer and winter steelhead were not listed. Most hatchery winter steelhead enter the river from November through January while wild winter steelhead enter the river in March through May.

The purpose of this study was to determine if steelhead smolts released downstream of the main juvenile chinook rearing area on the Lewis River altered the adult steelhead return distribution or impaired hatchery collection of adequate brood stock.

The Lewis River, located in southwest Washington State, flows for 31.4 km below Merwin Dam, the upstream terminus for anadromous fish, before entering the lower Columbia River at rkm 140 (Fig. 1). Merwin Hatchery, the rearing location of steelhead in this experiment is located at rkm 30.6. Lewis River Salmon Hatchery is located at rkm 25.3.

At Merwin Hatchery in September 1995 and 1996, four groups of about 60,000 each of winter and summer steelhead were adipose fin-clipped and tagged with blank magnetic wire tags in the left or right cheek and placed in one of four ponds. Left-cheek tagged winter steelhead were placed in one pond, left-cheek summer steelhead in a second pond, right-cheek tagged winter steelhead in a third pond and right-cheek summer steelhead in the fourth pond. The fish were reared in identical 56.1 m x 25.6 m x 1.4 m asphalt-lined ponds with a water flow of 950 l/min until release in mid April. Fish were fed *ad libitum* with demand feeders.

On the day prior to smolt release, 300 fish from each pond were measured for fork length to the nearest mm. In addition, 100 fish were weighed to the nearest g to determine Fulton's condition factor (weight(g)*100/(length(cm)^3). Tag retention was determined by passing 300 fish per pond through a wire-tag field sample detector (Northwest Marine Technology, Shaw Island, Washington[2]). The resulting retention rate (90-99%) was applied to the release number to determine the number of tagged fish released. Student t-test ($\alpha = 0.05$) was used to compare lengths for upper and lower releases for each steelhead race for both years of release.

Steelhead smolts were trucked downstream of Merwin Hatchery and released near the Lewis River Salmon Hatchery (rkm 25.3, upper release site) or about 1.6 km upstream of the town of Woodland at rkm 10.6 (lower release site). The upper and lower release sites were upstream and downstream of the main wild chinook rearing area, respectively.

The pond design at Merwin Hatchery allowed fish from two hatchery rearing ponds to enter one of two collection basins. Thus, winter and summer steelhead destined for

[2]Use of trade names does not imply endorsement by the California Department of Fish and Game.

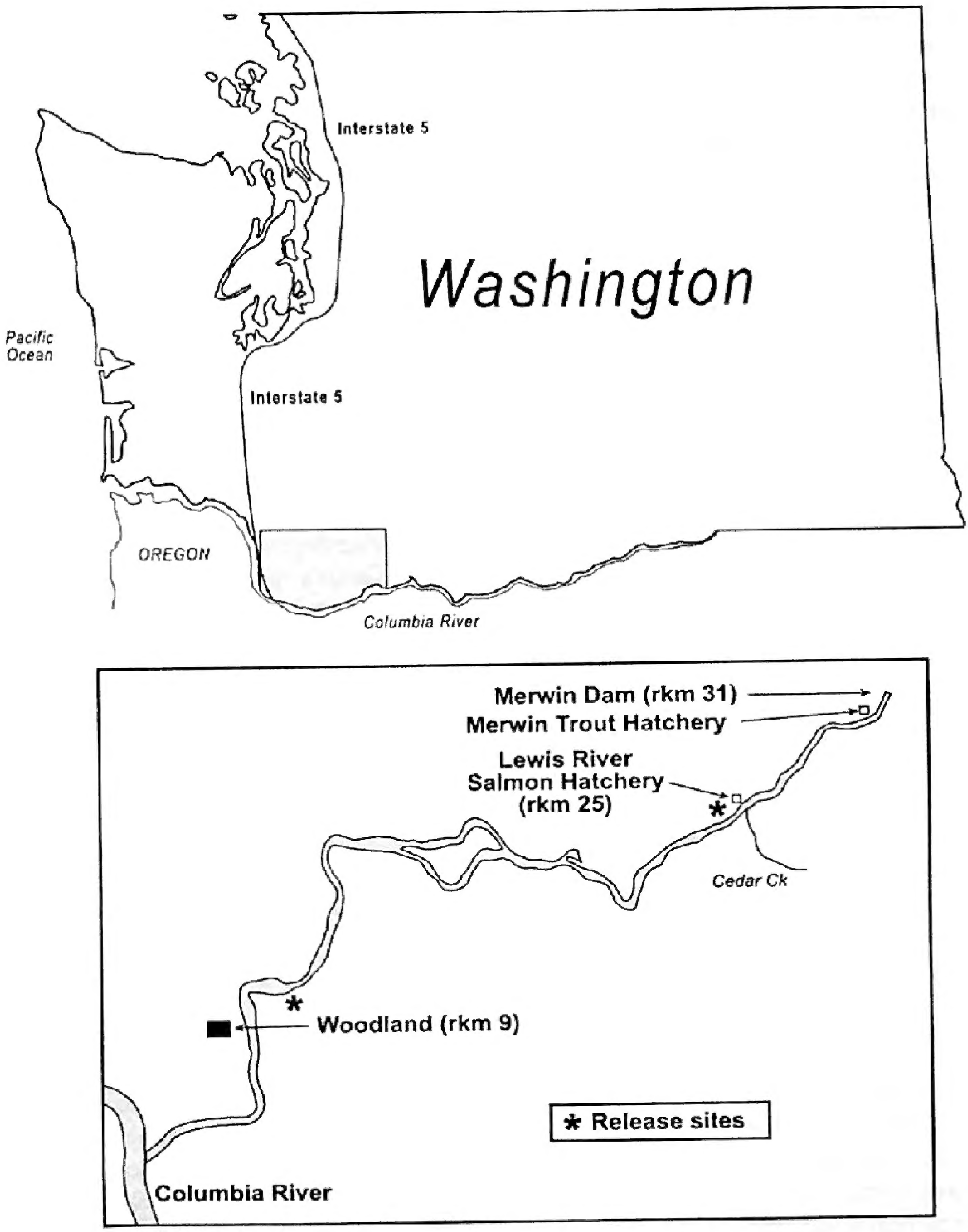

Figure 1. Release sites of hatchery steelhead smolts and adult recovery areas on the Lewis River, Washington.

the lower-river site (left-cheek tagged) were gathered concurrently in one collection basin while fish from the other two rearing ponds, destined for the upper-river site (right-cheek tagged), were gathered in an adjacent collection basin. Truckloads of fish were taken alternately from each collection basin to their designated sites. Releases were completed within a week.

Tags from adult steelhead were detected with a wand detector (Northwest Marine Technology, Shaw Island, Washington) which indicated the presence of a tag in the left or right cheek, similar to Tipping and Heinricher (1993). Briefly, the wand was rubbed on both cheeks of the fish and because the wand detector has a range of about 2.5 cm, the cheek containing the tag was readily identified. Tagged fish from each detection were recorded by date and location.

Adult steelhead were recovered at the hatchery rack at Lewis River Salmon Hatchery, a v-weir trap at Merwin Dam, and a v-weir trap in the fish ladder near the mouth of Cedar Creek (rkm 25.0). Because of the close proximity of the Cedar Creek trap to Lewis River Salmon Hatchery, tagged fish recovered at Cedar Creek trap were included with the Lewis River Salmon Hatchery recoveries.

Adult steelhead from each release were recovered as 2-year ocean fish and 3-year ocean fish. In winter 1998-1999 and summer 1999, 2-year ocean steelhead from the 1997 release and 3-year ocean fish from the 1996 release returned to the river concurrently. In that year, length frequency was used to apportion tagged fish to each release; fish over 73 cm fork length were allocated to the 1996 release and fish less than or equal to 73 cm were allocated to the 1997 release. This length criteria was based on length frequencies of tagged 2-year ocean fish returning in winter 1997-1998 and summer 1998. Although a few fish may have been apportioned to the wrong release year due to overlapping length frequency, the same side cheek was tagged in both years for each release location and thus the two-year total recoveries would correctly identify release location.

The Lewis River Salmon Hatchery trap was operated from mid-September to late December each year to collect salmon brood stock. The trap at Merwin Dam was operated from mid April through January to gather coho salmon, *O. kisutch*, spring chinook salmon and steelhead brood stock. Steelhead were checked for tags, marked with a hole punch in an operculum or by removing a small portion of the upper caudal fin so they would not be counted a second time, and then released to the river for additional sport-fishery opportunity or taken to Merwin Hatchery for brood stock. Fish captured at Merwin Dam that were not retained for brood stock were released at rkm 18.8 and 2.9. Fish recovered at Lewis River Salmon Hatchery were released to the river at that site.

The trap at Cedar Creek was fished year-round beginning February 1998. Trapped hatchery steelhead were checked for tags and marked with anchor tags to avoid double counting and released back to the river at rkm 24.0.

Adult return distribution to Merwin Dam and Lewis River Hatchery was standardized to 60,000 smolts per year per release site and then compared using a 2 x 2 Chi-square test (α= 0.05) for each release year. Recoveries for each release year were also compared for each race of steelhead and if not significantly different, years were combined and

compared again. The null hypothesis was that there was no difference in recovery distribution by release site.

Although smolt length differences for upper and lower releases were less than 1 cm for each year and sometimes differed by only 0.3 cm, the differences were significant for each release (Table 1). However, the study focus was on proportional returns by release site and we assumed that smolt release lengths did not influence adult return distribution. In addition, release lengths were thought to be biologically similar based on steelhead smolt survival studies (Tipping et al. 1995, Tipping 1997).

Table 1. Length (cm) and condition factor (K) of steelhead released at upper and lower river sites. Lower-release-site fish were tagged with blank wire in the left cheek while upper-release-site fish were tagged with blank wire in the right cheek. Sample size for lengths and K was 300 and 100, respectively. An asterisk next to a length indicates a significant difference for the lower versus upper release site.

Release year	Stock	Release site	Mean length (SD)	Mean k (SD)	Release number
1996	Summer	Lower	20.1 (1.6)*	0.92 (0.06)	55,268
		Upper	20.8 (1.5)	0.94 (0.08)	58,001
	Winter	Lower	20.2 (1.2)**	0.94 (0.06)	54,213
		Upper	20.7 (1.3)	0.95 (0.06)	60,580
1997	Summer	Lower	18.9 (1.1)***	1.02 (0.05)	59,934
		Upper	19.2 (1.0)	1.02 (0.05)	60,843
	Winter	Lower	20.0 (1.3)****	1.01 (0.07)	59,740
		Upper	19.7 (1.4)	1.04 (0.06)	60,047

* $t=5.53$, $df=598$, $p<0.005$
** $t=4.90$, $df=598$, $p<0.005$
*** $t=3.50$, $df=598$, $p<0.005$
**** $t=2.72$, $df=598$, $p<0.005$

Trap recoveries of tagged winter steelhead totaled 740 fish, 357 from the lower release and 383 from the upper release site (Table 2). A total of 63.9% of lower release site winter steelhead recoveries were obtained at Lewis River Salmon Hatchery compared to 53.3% of upper-river released fish. Conversely, 36.1% of the lower-river released fish were recovered at Merwin Dam compared to 46.7% of upper-river released fish. This difference was significant for the 1996 release ($\chi^2 = 7.29$, $df = 1$, $P = 0.007$), not significantly different for the 1997 release ($\chi^2 = 2.38$, $df = 1$, $P = 0.123$), and significantly different when both releases were combined ($\chi^2 = 8.17$, $df = 1$, $P = 0.004$).

Table 2. Hatchery rack recoveries (%) of winter steelhead in the Lewis River by release site and release year. Lewis River Hatchery is located at rkm 25 and Merwin Dam is located at rkm 31.

Release Site	Release Year	Lewis R. Hatchery	Merwin Dam	N
Lower	1996	101 (57.7)	74 (42.2)	175
	1997	127 (69.8)	55 (30.2)	182
	Total	228 (63.9)	129 (36.1)	357
Upper	1996	82(43.9)	105(561)	187
	1997	122(62.2)	74(37.8)	196
	Total	204(53.3)	179(46.7)	383

Trap recoveries of tagged summer steelhead totaled 1,861 fish, 871 from the lower release site and 990 from the upper release site (Table 3). Adult recovery distribution of the two groups was similar, the difference being about 1%. The difference was not significantly different for the 1996 release ($\chi^2 = 0.47$, df = 1, P = 0.494) or 1997 release ($\chi^2 = 0.15$, df = 1, P = 0.697). Summer steelhead releases were not combined because there were significant differences in recovery rates between release years.

Table 3. Hatchery rack recoveries (%) of summer steelhead in the Lewis River by release site and release year. Lewis River Hatchery is located at rkm 25 and Merwin Dam is located at rkm 31.

Release site	Release year	Lewis R. Hatchery	Merwin Dam	N
Lower	1996	106 (19.3)	442 (80.7)	548
	1997	4 (1.8)	220 (98.2)	224
	Total	110 (12.6)	662 (87.4)	871
Upper	1996	110 (17.5)	519 (82.5)	629
	1997	5 (1.4)	356 (98.6)	361
	Total	115 (11.6)	875 (88.4)	990

Our fish trapping results showed mixed (winter steelhead) and lack (summer steelhead) of release site fidelity, and contrasts to results of Wagner (1969), Slaney et al. (1993) and Cramer (1981). Relatively short river length and distance between release sites may be key to the difference. The streams in Wagner's (1969) and Cramer's (1981) studies were two to three times longer than the Lewis River. Slaney et al. (1993) found small but significant release-site fidelity differences on a stream of similar length as the Lewis River but also observed substantial dispersal of lower-river-released fish to the upper river near the rearing hatchery. If our planting sites would have been separated

by a greater distance, perhaps release-site fidelity would have been enhanced. Transported smolts are known to home as adults to the release location on streams without a hatchery (Kenaston et al. 2001) but presence of a rearing hatchery upstream of release sites may overshadow adult redistribution attempts of downstream releases, especially on relatively short streams with summer steelhead. Survival of endangered wild chinook salmon may have been enhanced by transporting steelhead smolts below the main juvenile rearing area. However, additional research is needed to determine the benefit and costs of transporting smolts downstream versus the number of wild chinook fry saved from smolt predation, and if those fry would have survived to adults.

The increased release-site fidelity for winter steelhead over summer steelhead in our study may be due to the relatively short time interval between river entry and spawning; hatchery winter steelhead typically enter the river in late November through December and spawn at the hatchery in December and January. Summer steelhead enter the river in May through July but do not spawn until December and January, allowing considerable time to wander.

Releasing smolts in the lower river did not impair the ability to collect adequate brood stock. Only about 50 summer steelhead females and 30 winter steelhead females are needed for the hatchery program and trap recoveries far exceeded this need.

ACKNOWLEDGMENTS

We thank Laurie Fowler, Joe Banks, Wolf Dammers and Frank Shrier for manuscript review. R. Nicolay and the hatchery staff greatly assisted with hatchery rack recoveries. D. Pruett produced the graphics and P. Hahn helped with the statistical analysis. This study was funded by PacifiCorp, Portland, Oregon.

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INCIDENCE OF WHITE STURGEON DEFORMITIES IN TWO REACHES OF THE COLUMBIA RIVER

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We sampled 2 functionally isolated populations of white sturgeon, *Acipenser transmontanus*, to measure the occurrence of physical deformities. In the Columbia River estuary we found the frequency of all deformities among white sturgeon to be 8.1%, whereas in The Dalles Reservoir the frequency was 11.7%. The most common deformity observed in white sturgeon from the estuary was an additional row of lateral scutes on both sides of the fish. In The Dalles Reservoir the most prevalent malformation was misshapened fins, typically seen as bilateral curled pectoral fins, followed by abnormal barbels manifested as shortened or forked barbels. The frequency of misshapened fins was significantly greater among white sturgeon in The Dalles Reservoir than in the estuary. In the estuary, the rate of deformities increased with size. Although no causal relationship has been determined, Columbia River sediment is contaminated with organic pollutants that are known to be harmful to aquatic organisms. Further study is needed to evaluate whether the observed deformities have an environmental or genetic basis.

INTRODUCTION

Morphological anomalies are uncommon in natural fish populations (Dahlberg 1970), yet deformed wild sturgeon have been reported. Schwartz (1992) described a Russian beluga sturgeon, *Huso huso*, possessing spinal and body deformities. Dadswell et al. (1984) summarized abnormalities observed in shortnose sturgeon, *Acipenser brevirostrum*, during 6 years of sampling in the St. John Estuary, Canada. Deformities included nasal septum absence, spinal curvature, blindness, and extra or missing fins. We surveyed sturgeon biologists and found that deformities, especially those affecting the fins, have been observed in Atlantic sturgeon, *A. oxyrhynchus*, captured from the Hudson River, New York (Jerre Mohler, U.S. Fish and Wildlife Service, personal communication); lake sturgeon, *A. fulvescens*, from the Moose River, Ontario (David Noakes, University of Guelph, personal communication); and white sturgeon, *A. transmontanus*, from several British Columbia waters including the upper Columbia River (Larry Hildebrand, RL&L Environmental Services Ltd., personal communication) and the Nechako and Fraser river drainages (Scott McKenzie, RL&L Environmental Services Ltd., personal communication) and from the Sacramento-San

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Joaquin Estuary, California (David Kohlhorst, California Department of Fish and Game, personal communication).

Previous studies implicated water-borne contaminants as 1 agent contributing to abnormalities in wild fish populations (Mehrle et al. 1982, Malins et al. 1984, Lindesjöö and Thulin 1992). Several water-borne contaminants in natural systems are found to occur in river sediments (CBFWA[2] 1996). In the lower Columbia River Basin, there is strong evidence that fish and wildlife are being exposed, via water, sediments, and prey, to a range of pollutants, including heavy metals, organochlorine pesticides, dioxins and furans, and other organic compounds known to cause adverse physiological effects (Tetra Tech[3] 1996). Because white sturgeon are primarily bottom-dwelling organisms that commonly feed on benthos, they are regularly exposed to sediment-borne contaminants. Therefore, a causal link between contaminant concentrations and white sturgeon deformity in the Columbia River may exist. Other factors contributing to the frequency of deformity among fishes include nutritional deficiencies (Halver et al. 1969, Lovell and Lim 1978), environmental variables such as water temperature, salinity, and dissolved oxygen concentration (Seymour 1959, Alderdice and Velsen 1971, Rosenthal and Alderdice 1976), disease (Sindermann 1979), injury (Dahlberg 1970) and genetic disposition (Gordon 1954).

White sturgeon roamed freely throughout the Columbia River Basin prior to the introduction of hydropower development in 1933 with the construction of Rock Island Dam. During the next 35 years, 10 additional dams were constructed on the mainstem Columbia River. Although some impounded white sturgeon migrate past Columbia River dams via fish ladders and locks (Warren and Beckman[4] 1993), most exhibit restricted movement and likely remain in a single reservoir their entire life (ODFW and WDFW[5] 1998). The result is a series of functionally isolated white sturgeon populations throughout the Columbia River basin (North et al. 1993). Each discrete population is subjected to dam operations that regulate seasonal discharge patterns and reservoir retention times, thus affecting water temperature, depth, and nutrient and contaminant loading. Additionally, point source contaminants vary among reaches and potentially impact reservoir white sturgeon populations disproportionately.

We took advantage of ongoing studies to compare the frequency of deformity between an impounded population of white sturgeon and an unimpounded, diadromous population. Previous field observations led us to suspect a high incidence of deformity among Columbia River white sturgeon. In 1997, we began monitoring white sturgeon

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[2]CBFWA (Columbia Basin Fish and Wildlife Authority). 1996. Contamination ecology of selected fish and wildlife of the lower Columbia River. Prepared for Lower Columbia River Bi-State Program. CBFWA, Portland, Oregon, USA.

[3]Tetra Tech. 1996. The health of the river, 1990-1996. Prepared for Lower Columbia River Bi-State Program. Tetra Tech, Inc., Redmond, Washington, USA.

[4]Warren, J. J. and L. G. Beckman. 1993. Fishway use by white sturgeon on the Columbia River. Washington Sea Grant Program, Seattle, Washington, USA.

[5]ODFW (Oregon Department of Fish and Wildlife) and WDFW (Washington Department of Fish and Wildlife). 1998. Status report: Columbia River fish runs and fisheries, 1938-97. ODFW, Portland, Oregon, USA.

by examining individuals for growth deformities. Relative health of Columbia River white sturgeon from an impounded reach and from the estuary was documented by recording the presence or absence of deformities.

METHODS

The study area included 2 regions of the mainstem Columbia River: The Dalles Reservoir (river km [rkm] 309-347) and the upper estuary (rkm 30-45) (Fig. 1). The Dalles Reservoir (Lake Celilo) is a 4,500-ha impoundment formed in 1957 by the construction of The Dalles Dam. The 2nd area included a 15-km section of the estuary in the lower Columbia River, a free-flowing stretch providing direct access to the ocean. Two dams and approximately 264 rkm separate these study areas.

White sturgeon were examined for morphological deformities in conjunction with ongoing white sturgeon tagging studies (North et al. 1999[6], Watts and Whisler[7] 1998).

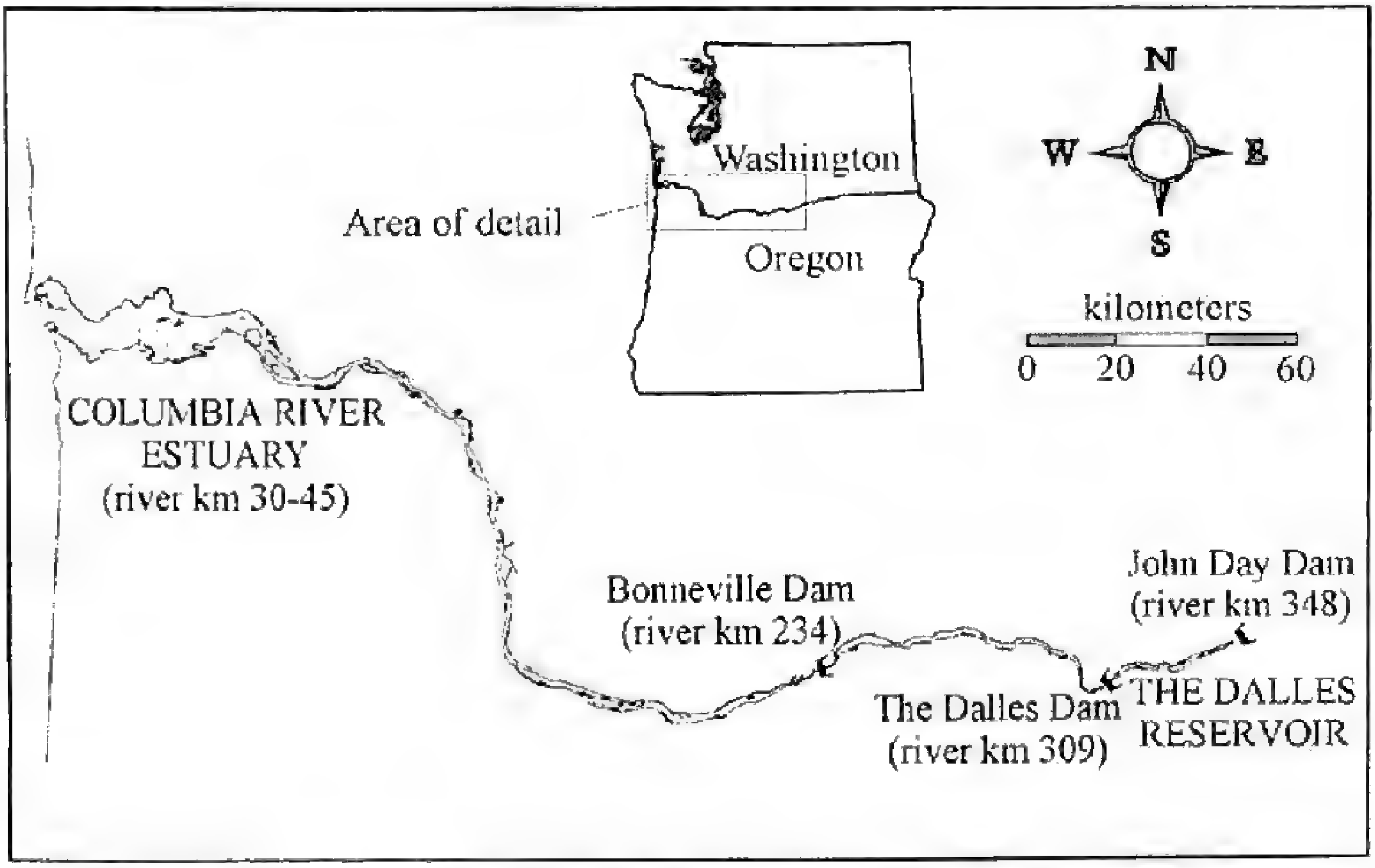

Figure 1. White sturgeon deformity study sites (all capital letters) on the Columbia River, 1997. The scale is approximate.

[6]North, J. A., L. C. Burner, and R. A. Farr. 1999. Report A *in*: D. L. Ward, editor. Effects of mitigative measures on productivity of white sturgeon populations in the Columbia River downstream from McNary Dam, and status and habitat requirements of white sturgeon populations in the Columbia and Snake rivers upstream from McNary Dam. Annual Progress Report to Bonneville Power Administration, Portland, Oregon, USA.

[7]Watts, J. W. and J. Whisler. 1998. The 1997 NEAP Funded sturgeon tagging project on the lower Columbia River. Oregon Department of Fish and Wildlife. Annual Report to Pacific States Marine Fisheries Commission, Portland, Oregon, USA.

These large-scale efforts provided sample sizes that enabled us to measure the prevalence of deformities among white sturgeon. In The Dalles Reservoir, white sturgeon were captured using setline gear. Setlines were 183-m long with 40 circle hooks attached every 4.6 m. Hook sizes were 10/0, 12/0, 14/0, and 16/0 and were baited with pickled squid, *Loligo* spp. Lines were fished for an average 23 hours per set. Multifilament nylon nets (19.7-cm stretched mesh) were used to capture white sturgeon in the Columbia River estuary. Nets measured 274-366 m in length by 10 m in depth. Nets were fished for approximately 1 hour.

A numerical coding system was developed to aid samplers in recording deformities. Single digit codes were assigned to each deformity type. Multiple digits indicated the presence of more than one deformity. Each white sturgeon was assigned a code after visual examination of all sides of the fish. No attempt was made to quantify the degree of individual deformities; we merely documented presence or absence.

Separate field crews using different gear types sampled each area. Deformity criteria applied among crew members and between crews may exhibit inherent differences. Therefore, care should be exercised when using this information since the current rating system is subjective. In addition, our analyses assumed both gear types had equal likelihood of capturing deformed fish.

The frequency of fish with misshapened fins was compared between the estuary and The Dalles Reservoir using a chi-square test of independence, as was the frequency of all other deformities combined. Differences in deformity frequencies were considered statistically significant when $P < 0.05$. In addition, we compared our results to similar data summarized by Macy et al.[8] (1997) from the John Day Reservoir. A chi-square contingency table was used to test for independence of deformity frequency among reaches.

Fisher's exact test (FET) of independence was used to compare the rate of deformity among 10-cm fork length (FL) intervals in white sturgeon by region. Correlation analyses were performed on fish length and rate of deformity.

RESULTS

We examined 1,459 white sturgeon during July and August 1997 in The Dalles Reservoir, of which 171 (11.7%) had at least 1 growth deformity (Table 1). The most prevalent malformation was misshapened fins, typically seen as bilateral curled pectoral fins, followed by abnormal barbels manifested as shortened or forked barbels. Multiple deformities were apparent in a few individuals ($n = 16$). The most common combination, observed in 6 individuals, was misshapened fins coupled with abnormal barbels.

Of 383 fish examined in the Columbia River estuary during 3 days of sampling in August 1997, 31 (8.1%) exhibited a physical deformity (Table 1). The most common

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[8]Macy, T. L., C. L. Burley, and W. Ambrogetti. 1997. Sturgeon studies of the John Day Reservoir, on the Columbia River, 1979-1981. U.S. Fish and Wildlife Service, Columbia River Fisheries Program Office, Vancouver, Washington, USA.

Table 1. Incidence of white sturgeon deformities in The Dalles Reservoir and the Columbia River estuary, 1997.

	The Dalles Reservoir		Columbia River estuary	
Deformity	Number observed	Percent of total	Number observed	Percent of total
No anomaly	1,288	88.3	352	91.9
Atypical skin coloration	4	0.3	5	1.3
Misshapened fin(s)	103	7.0	6	1.6
Total or partial fin absence	4	0.3	3	0.8
Eroded nare(s)	9	0.6	1	0.3
Abnormal barbel(s)	19	1.3	0	0.0
Asymmetrical snout	3	0.2	0	0.0
Eye(s) malformed or missing	3	0.2	2	0.5
Skeletal deformity	4	0.3	0	0.0
Other	6[a]	0.4	14[b]	3.6
Fish with multiple anomalies	16	1.1	0	0.0
Total	1,459	100.0	383	100.0

[a]Two fish with incomplete opercles; 1 fish with a fused pelvic fin; and 3 fish with tumorous growths.

[b]Thirteen fish with extra rows of scutes and 1 fish with an incomplete opercle.

deformity (13 fish, 3.4%) observed was an additional row of lateral scutes on both sides of the fish. Misshapened fins and atypical coloration were the next most frequently observed abnormalities.

The frequency of misshapened fins was significantly greater among white sturgeon in The Dalles Reservoir than in the estuary ($X^2 = 16.44$; df = 1; $P < 0.001$). Analysis of deformities other than misshapened fins revealed no significant difference in frequency between the 2 reaches ($X^2 = 2.205$; df = 1; $P = 0.138$). Comparisons of deformity frequency among the estuary, The Dalles Reservoir, and John Day Reservoir showed significant differences among reaches ($X^2 = 48.481$; df = 2; $P < 0.001$).

The rate of deformity varied significantly among FL intervals in white sturgeon from The Dalles Reservoir (FET: df = 12; $P < 0.001$); however, correlation analyses found no association between FL and incidence of deformity (Fig. 2). In the estuary, the rate of deformity was significantly different among FL intervals (FET: df = 8; $P = 0.006$) and correlation analyses showed an association between FL and incidence of deformity.

DISCUSSION

The occurrence of abnormalities observed in white sturgeon from the Columbia River far exceeds rates reported for other fish populations. A literature review by Berra and Au (1981) cited 6 studies where the frequency of deformities among non-

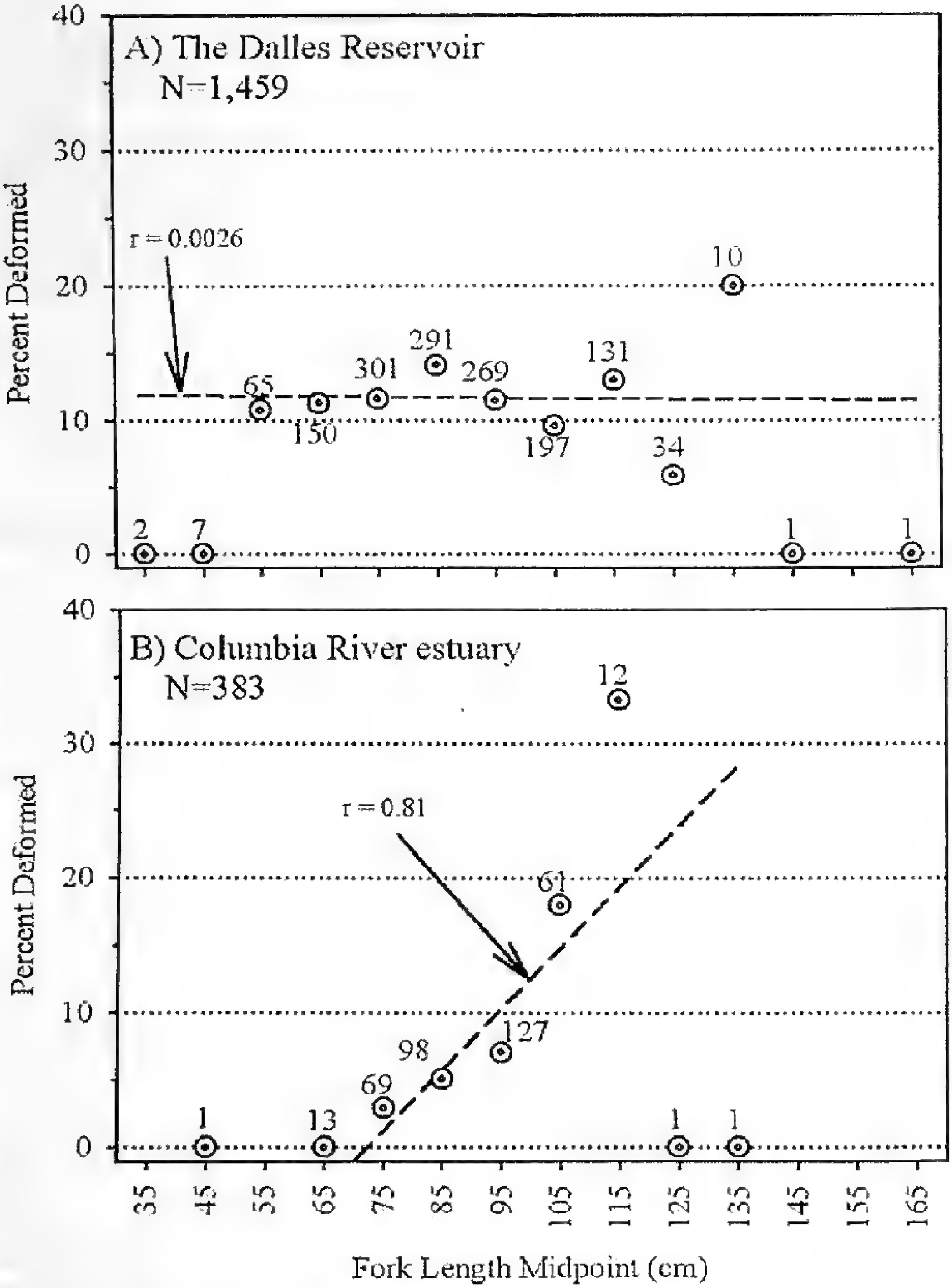

Figure 2. Percent deformed white sturgeon by fork length midpoint in A) The Dalles Reservoir and B) Columbia River estuary, 1997. Sample size for each fork length interval is indicated.

acipenserid fish populations ranged from 0.03% to 0.24%, with a weighted average of 0.18%. Their own study revealed 47 deformed individuals (0.26%) out of 18,361 fishes collected from Cedar Fork Creek, Ohio. This compares to our frequency of 8.1% to 11.7% for the 2 study populations.

Previous Columbia River white sturgeon studies have reported physical deformities. Bajkov (1955) noted white sturgeon with 7 rows of scutes while tagging in the Columbia River near Prescott, Oregon (rkm 114). Approximately 0.3% of nearly 7,000 fish possessed this anomaly. In John Day Reservoir (Lake Umatilla), located immediately upriver from The Dalles Reservoir, 5.8% of 2,965 white sturgeon exhibited abnormalities (Macy et al.[8] 1997). Similar to the results from The Dalles Reservoir, misshapened fins comprised the majority of the deformities in this impoundment. Anders and Beckman[9] (1993) found that a greater percentage of juvenile white sturgeon in The Dalles Reservoir (4-18%) were physically deformed than in Bonneville (Lake Bonneville) (0-0.5%) or John Day (0%) reservoirs. Again, fin deformities were the most prevalent physical anomaly.

Differences in deformity frequency among the estuary, The Dalles Reservoir, and John Day Reservoir were probably driven by the relatively high rate of misshapened fins in the reservoir populations. This is consistent with findings of previous white sturgeon studies discussed above.

Deformity incidence of white sturgeon in both study areas varied with FL, but only the estuary showed a positive correlation between fork length and rate of deformity. In The Dalles Reservoir, small numbers of large fish with no deformities may have masked an otherwise observable increase in deformity rate with size. Our observations found that larger fish exhibited greater rates of deformity. This may be attributed to older individuals having experienced more injurious events or long-term exposure to environmental insults having cumulative effects.

Gross morphological defects in aquatic organisms have been associated with contaminant exposure (Warwick et al. 1987, Hamilton and Reash 1988, Fournie et al. 1996) but without demonstration of a cause-and-effect relationship. No concurrent sediment or fish tissue sampling was conducted. Thus, we can only speculate on what may be contributing to the high rate of deformity among Columbia River white sturgeon but we believe contaminant exposure to be a likely agent. In 1990 the Lower Columbia River Bi-State Water Quality Program was initiated to study water quality from Bonneville Dam to the Pacific Ocean (Fuhrer et al. 1996). Analyses of river water, sediment, and biota from the lower river suggest that aquatic organisms and wildlife are being exposed to heavy metals and organochlorine chemicals such as pesticides and polychlorinated biphenyls (PCBs), polychlorinated dibenzodioxins (dioxins), and

[9]Anders, P. J. and L. G. Beckman. 1993. Comparisons of white sturgeon egg mortality and juvenile deformity among four areas of the Columbia River. Pages 355-363 *in*: R. C. Beamesderfer and A. A. Nigro, editors. Status and habitat requirements of the white sturgeon populations in the Columbia River downstream from McNary Dam, Volume 2. Final Report (contract DE-AI79-86BP63584) to Bonneville Power Administration, Portland, Oregon, USA.

polychlorinated dibenzofurans (furans) throughout the Columbia River basin (Tetra Tech[3] 1996).

Heavy metals enter the lower Columbia River via stormwater runoff, sewage treatment facililities, and mining and manufacturing processes. A point source study identified 12 chemical plants, 11 wood products facilities, 4 aluminum plants, and 29 sewage treatment plants contributing to contaminant loading in the lower Columbia River (Fuhrer et al. 1996). Sources upstream of Bonneville Dam are also believed to contribute to the elevated concentrations of heavy metals observed in the lower river. The Bi-State study found that heavy metals, including lead, mercury, copper, selenium, aluminum, and iron, exceeded federal ambient water-quality criteria. Laboratory studies have determined that heavy metals may act as teratogens in fish. Lemly (1997) found teratogenic deformities of the skeleton, head, fins, and mouth to be reliable bioindicators of selenium poisoning in fish. Similar deformities were found in fathead minnows, *Pimephales promelas*, where the percentage of larvae with abnormalities increased significantly with increasing copper concentration (Scudder et al. 1988). Mercury is reported to causal spinal curvatures in killifish, *Fundulus heteroclitus* (Weis and Weis, 1977). Demersal organisms such as white sturgeon may be at increased risk to heavy metal exposure as these substances accumulate in sediment.

Organochlorine chemicals are ubiquitous environmental contaminants that tend to biomagnify in aquatic food chains because of their resistance to degradation and high lipophilicity (Noguchi 1998). Sturgeon are particularly susceptible to organochlorine contamination due to their high body fat content (Mecozzi[10] 1988). Additionally, the longevity of white sturgeon allows for greater accumulation of toxic substances due to repeated or chronic exposure. The most documented organochlorine pesticide in the Columbia River is dichlorodiphenyl-trichloroethane (DDT). Although banned in the early 1970s, DDT and its metabolites persist and continue to be detected in tissue from Columbia River fish and wildlife. Foster et al. (2001) detected DDT or one of its metabolites in gonadal tissue from every white sturgeon sampled in The Dalles Reservoir (n=11) and the Columbia River Estuary (n=7). We urge caution in implicating DDT exclusively because few studies have identified organochlorine pesticides as direct agents of deformity. One exception is Couch et al. (1977) who found Kepone induced scoliosis in sheepshead minnows, *Cyprinodon variegatus*.

Organochlorines of special concern are PCBs because they are known mutagenics and teratogenics (Bosley and Gately[11] 1981, Sanderson et al. 1994). Although the manufacture of PCBs was discontinued in 1977, Columbia River biota continues to be impacted by their environmental persistence. Furthermore, PCBs continue to enter the Columbia River system via older hydroelectric transformers that utilize PCBs as

[10]Mecozzi, M. 1988. Lake sturgeon (*Acipenser fulvescens*). Wisconsin Department of Natural Resources Bureau of Fisheries Management, PUBL-FM-704 88, Madison, Wisconsin, USA.

[11]Bosley, C. E. and G. F. Gately. 1981. Polychlorinated biphenyls and pesticides in Columbia River white sturgeon (*Acipenser transmontanus*). U.S. Fish and Wildlife Service, Marrowstone Field Station, Nordland, Washington, USA.

insulators and lubricants (personal communication with John G. Wegrzyn and Matt McClincy, Northwest Region, Oregon Department of Environmental Quality). Investigators have determined that Columbia River fish exhibited high levels of PCBs that in some cases exceeded national standards (Anthony et al. 1993, TetraTech[3] 1996).

Dioxins and furans are an extremely toxic group of organochlorine chemicals. They are released into the Columbia River primarily through chlorine bleaching of pulp and paper conducted at 5 mills below Bonneville Dam (Fuhrer et al. 1996). Analyses of tissue from largescale suckers, *Catostomus macrocheilus*, and white sturgeon from the lower Columbia River found that dioxin and furan levels exceeded contaminant burden reference levels used by the Bi-State Water Quality Program (Tetra Tech[3] 1996). A study in Sweden suggested that toxic substances in bleached kraft mill effluent from 3 pulp mills contributed to skeletal deformities in northern pike, *Esox lucius* (Lindesjöö and Thulin 1992). Perhaps similar processes are impacting Columbia River white sturgeon.

Further study is needed to evaluate whether the observed deformities have an environmental or genetic basis. However, documentation of compromised health in Columbia River fish and wildlife and the expression of deformities among resident white sturgeon are powerful measures of exposure to environmental contaminants. Whether contaminants may be expressing teratogenic effects independently or synergistically has yet to be determined. Nevertheless, these data provide baseline information to document the incidence of deformity in Columbia River white sturgeon and help investigators monitor long-term trends that may require closer study.

ACKNOWLEDGMENTS

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RECORDS OF THE SHOKIHAZE GOBY, *TRIDENTIGER BARBATUS* (GÜNTHER), NEWLY INTRODUCED INTO THE SAN FRANCISCO ESTUARY

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The shokihaze goby, *Tridentiger barbatus* (Günther) (Fig. 1; formerly categorized as *Triænophorichthys barbatus* Günther, 1861 and *Triænopogon barbatus* Bleeker, 1874) was recently introduced into the San Francisco Estuary. The most likely mechanism of this introduction was the release of ballast water or eggs laid on fouling material on a ship hull. Ballast water has been implicated as the transport mechanism in the introduction of several invertebrate and fish species into California waters (Carlton 1985). Ballast water release appears to be the most likely mechanism for introduction into the San Francisco Estuary for the yellowfin goby, *Acanthogobius flavimanus* (Brittan et al. 1963), the chameleon goby, *T. trigonocephalus* (Raquel 1988), and the shimofuri goby, *T. bifasciatus* (Matern and Fleming 1995).

The shokihaze goby is native to Taiwan, Korea, China, Hong Kong, and Japan (Günther 1861, Jordan and Snyder 1902, Tomiyama 1936, Dôtu 1957, Lin et al. 1994, Ni and Kwok 1999). Shokihaze is a Japanese name derived from Shoki, a bearded

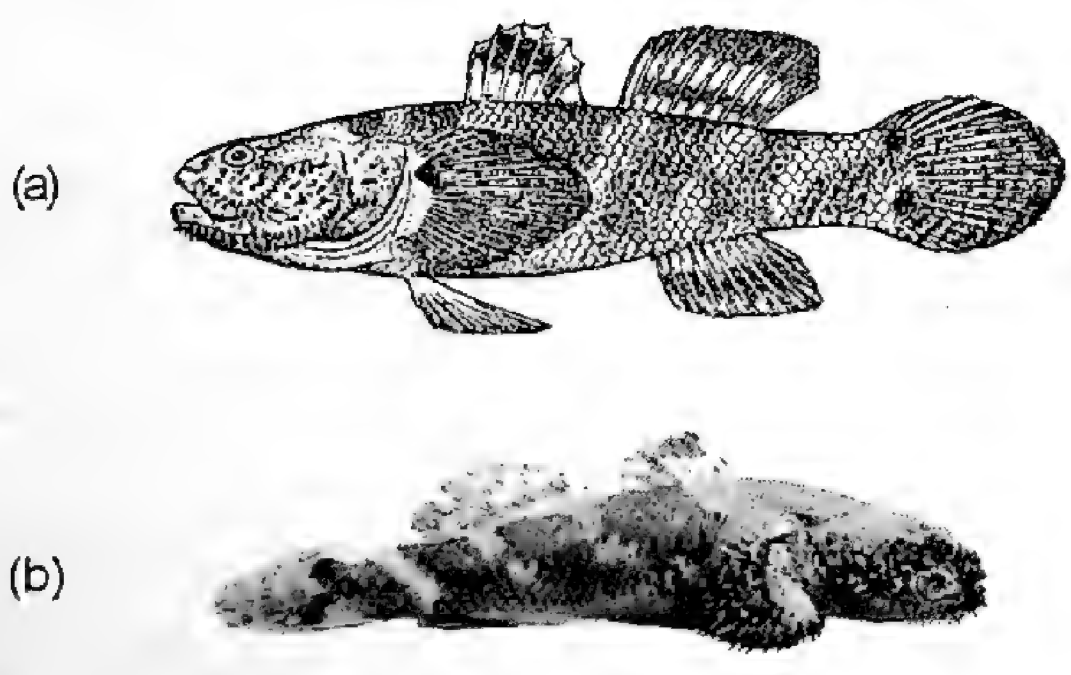

Figure 1. Shokihaze goby *Tridentiger barbatus* (Günther). (a) From a 112 mm TL (approximate) specimen. Modified by the author from Jordan and Snyder 1902. (b) From a 60 mm TL (approximate) specimen. Photograph by Lee Mecum, modified by the author.

mythological hero who wards off evil spirits (Minneapolis Institute of Arts[1]) and haze, which means goby. Goby is used after shokihaze to maintain consistency among the common names of gobies. The goby is named after Shoki because of the numerous short barbels found on the ventral margin of the suborbital, lower jaw, and preoperculum, which resemble a beard (Fig. 1).

The shokihaze goby superficially resembles a sculpin, except that the pelvic fins are modified into a suction cup and it possesses barbels. These characteristics along with two separate dorsal fins distinguish the shokihaze goby from all other fishes in California. Barbels are found on shokihaze gobies as small as 16 mm total length (TL) (Dôtu 1957), but on specimens < 20 mm TL the barbels are few in number and difficult to see even under magnification. The shokihaze goby reaches a maximum length of about 145 mm TL (Tomiyama 1936). When viewed from above, it has a wide head, which tapers sharply to the tail, resulting in a teardrop shape. The eyes are widely spaced compared to most native gobies. Coloration is also sculpin-like with a consistent pattern, which is evident at 18 – 20 mm TL. The shokihaze goby has a mottled appearance consisting of six dusky bands on a tan background: one at the nape, a second broad band over the back between the pectoral fins and four diagonal bands across the sides. The 3rd band starts just behind the pectoral fin and angles back to the middle of the 1st dorsal fin. The 3rd, 4th, and 5th bands are often connected along the side. The 6th band ends posteriorly at the caudal fin. The diagonal bands fade ventrally. There are two dark spots at the base of the caudal fin and dark pigmentation on the top and rear of the first dorsal fin.

Gobies of the genus *Tridentiger* can be identified by a row of trilobed teeth on both the upper and lower jaws (Masuda et al. 1984, Mukai et al. 1996). The shokihaze goby can be distinguished from the two other members of the genus introduced into California, the chameleon goby and the shimofuri goby, by the presence of barbels on the head, the diagonal bands on the sides and a sharper taper from head to tail. However, shokihaze gobies < 16 mm are difficult to differentiate from larval shimofuri gobies [L. Lynch, California Department of Fish and Game (CDFG), personal communication].

Three shokihaze gobies, captured in the San Francisco Estuary by the California Department of Fish and Game San Francisco Bay Study (Bay Study), have been deposited at California Academy of Sciences as voucher specimens (CAS 2000-x:13, 213527, 67 and 88 mm TL; and CAS 213528, 27 mm TL) and three shokihaze gobies have been deposited at Scripps Institution of Oceanography (SIO 00-51, 57 mm TL; SIO 01-120, 63 mm TL; SIO 01-121, 57 mm TL).

The first recorded capture of the shokihaze goby in California occurred 3 November 1997. It was caught with an otter trawl in the San Joaquin River at the Bay Study station 853 (Table 1, Fig. 2) during a monthly survey. Catch remained low (0 – 5/month) until September 2000, when 16 small shokihaze gobies were caught. This catch included the first shokihaze gobies collected that were < 25 mm TL. During October 2000, 58 shokihaze gobies were caught and only 3 were >20 mm TL. Shokihaze goby catch was low in the first half of 2001, but catches in the second half of the year increased sharply,

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[1]Minneapolis Institute of Arts http://www.artsmia.org/mythology/slide24.html/

Table 1. Shokihaze goby catch, lengths (mm total length), and associated water parameters of depth (m), salinity (‰), and temperature (°C) for fish captured in the San Francisco Estuary by the California Department of Fish and Game San Francisco Bay Study (November 1997 – December 2000). Salinity and temperature represent bottom conditions.

Capture Date	Station	Number/ Station	Length (mm)	Depth (m)	Salinity (‰)	Temp (°C)
11/03/97	853	1	32	13.1	1.78	17.6
12/03/97	323	3	32,37,43	6.7	26.10	13.9
02/09/98	345	2	25,39	25.9	17.24	12.4
03/04/98	323	5	48-61	4.9	2.80	11.6
09/30/98	430	4	29-44	5.5	n/a	n/a
11/03/98	432	1	97	10.7	14.07	15.4
12/01/98	430	4	44-56	5.2	2.02	12.3
01/07/99	432	1	34	8.2	8.46	7.6
05/17/99	447	5	37-63	2.2	3.03	15.7
06/08/99	447	1	50	3.7	7.42	16.8
07/07/99	427	1	63	9.1	14.83	19.8
07/07/99	432	2	78,79	9.8	13.00	19.9
08/31/99	430	1	32	2.4	7.42	20.2
01/24/00	323	1	34	7.3	26.08	11.3
02/08/00	429	1	52	2.4	0.43	11.3
02/08/00	430	1	57	4.8	0.28	11.2
03/07/00	430	2	66,72	3.7	0.23	11.5
04/04/00	430	4	27-39	3.6	2.33	14.9
05/09/00	430	2	49,50	4.6	n/a	n/a
06/14/00	432	2	67,84	10.7	11.4	19.2
07/11/00	429	1	83	7.3	11.2[a]	n/a
07/12/00	433	2	67,88	12.5	3.5	19.9
08/28/00	430	1	30	2.9	7.00	20.3
08/28/00	433	1	90	13.1	5.03	20.7
09/18/00	736	10	13-38	12.5	2.20	21.6
09/18/00	750	6	18-38	9.8	0.74	21.3
10/20/00	736	3	17-21	10.5	2.40	18.2
10/20/00	750	51	16–21	9.2	1.12	17.9
10/20/00	751	1	17	10.3	0.91	17.7
10/20/00	853	3	33,44,53	13.0	0.63	18.3
10/21/00	433	1	89	12.7	7.87	17.7
11/13/00	751	1	31	10.7	0.44	12.7
11/13/00	853	1	37	11.3	1.45	13.2
11/14/00	447	1	44	4.0	18.82	12.7
12/07/00	736	1	43	10.4	7.24	10.7
12/11/00	428	1	56	12.2	21.65	11.1
12/11/00	429	5	45-60	8.8	17.20	11.1
12/11/00	432	14	36-103	12.5	22.50	11.1
12/11/00	447	5	32-109	4.9	24.54	11.2
12/11/00	535	4	29-44	15.5	9.23	11.0
12/12/00	325	1	46	12.2	28.81	11.4

n/a data not available.
[a] Surface reading

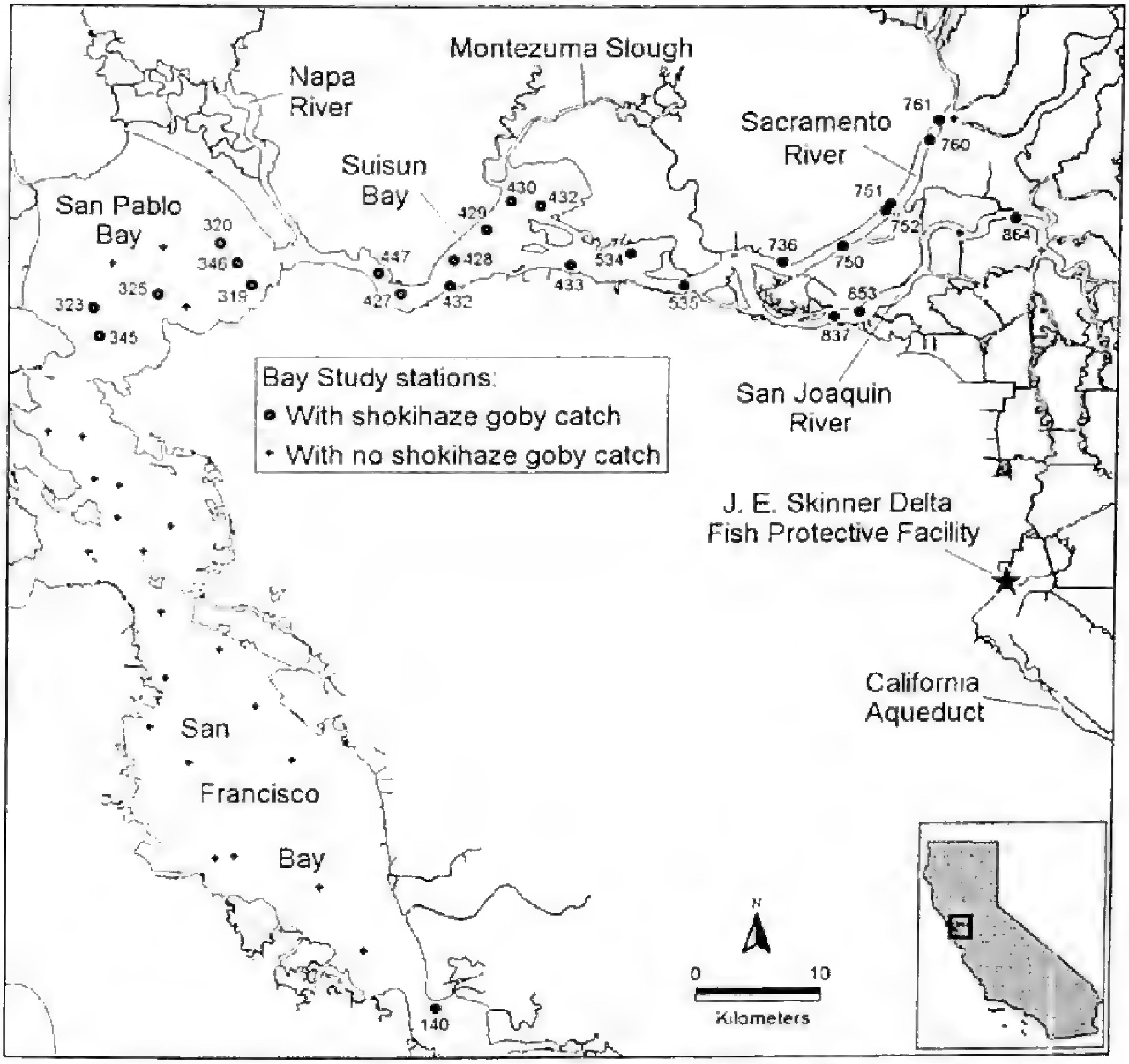

Figure 2. California Department of Fish and Game San Francisco Bay Study trawl stations with and without shokihaze goby catch (November 1997 through December 2001).

exceeding all prior catch by over 2 times (Fig. 3).

Shokihaze gobies have been caught by the Bay Study in otter trawl samples at stations from the Sacramento River near Rio Vista (Fig. 2, station 761) and the San Joaquin River near the mouth of the Mokelumne River (Fig. 2, station 864) downstream to south-western San Pablo Bay (Fig. 2, station 345) and southern San Francisco Bay (Fig. 2, station 140). Most large adults have been caught in the channels of Suisun Bay. Larvae and juveniles have been collected primarily in the lower Sacramento River.

Other studies have caught larvae in the Napa River (M. Dege, CDFG, personal communication) and Montezuma Slough (M. Bryant and K. Sousa, CDFG, personal communication). Shokihaze gobies have been caught by the Bay Study in depths from 2.2 to 25.9 m, in salinities ranging from 0.23 to 28.81 ‰ and at temperatures ranging from 7.6 to 21.6°C.

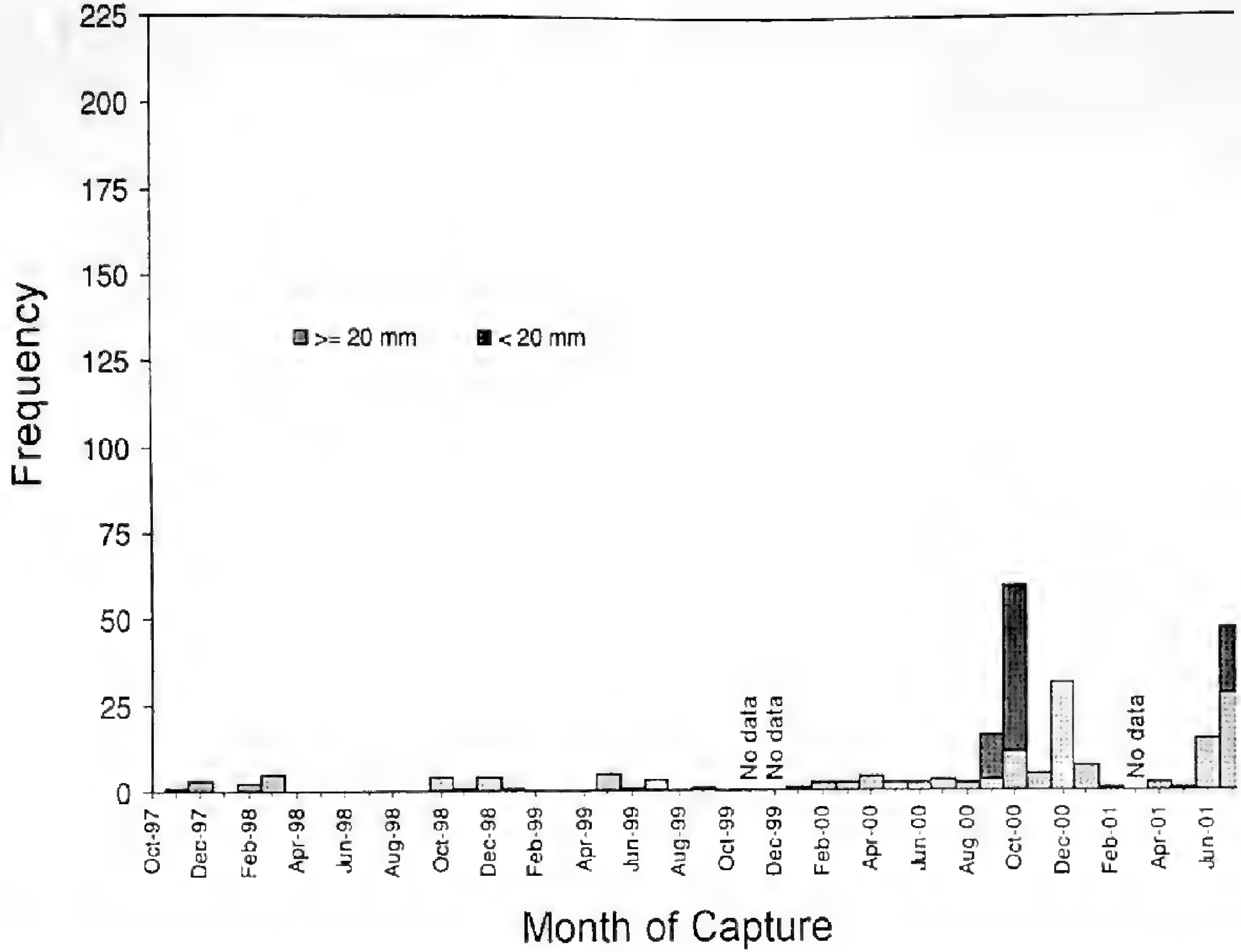

Figure 3. Shokihaze goby catch by month for two size classes (< 20 mm and = 20 mm TL) derived from otter trawl sampling by California Department of Fish and Game, San Francisco Bay Study (November 1997 through December 2001).

Little is known about the life history of the shokihaze goby in California. In Japan they are found in oyster, *Ostrea rivularis*, beds on mud flats (Dôtu 1957). Dôtu (1957) observed that they spawn in oyster shells from June through October. The male cleans and guards the eggs of one or more females until the eggs hatch. The eggs hatch after about 4 days in 25°C water and the young are planktonic until they reach about 18 mm TL. At this size, the young fish settle to the substrate and begin a demersal lifestyle.

Dôtu (1957) found small crustaceans, annelid worms, squid, and young fish, including shimofuri and shokihaze gobies, in the stomachs of the shokihaze goby.

Impacts of the shokihaze goby in California remain undetermined, but there is potential for negative impacts to native organisms, including listed species such as the federally endangered tidewater goby, *Eucyclogobius newberryi*, and the federally threatened delta smelt, *Hypomesus transpacificus*. Introduced fish however, usually do not cause the extirpation of native fish species (Moyle and Light 1996). Different habitat preferences for shokihaze gobies and both tidewater gobies and delta smelt should limit interactions between these fish. Tidewater gobies are most abundant in shallow water (< 2 m) habitat (Wang[2]), while the Bay Study has found that shokihaze gobies are most abundant in deeper water. The current ranges of the shokihaze goby and the tidewater goby do not overlap. The range of the shokihaze goby overlaps that of the delta smelt, but delta smelt are pelagic and adult shokihaze are demersal so there is limited opportunity for the shokihaze goby to prey upon the delta smelt (A. Rockriver, CDFG, personal communication).

The shokihaze goby may spread to southern California via the California Aqueduct. The shimofuri goby has already been transported to southern California through the California Aqueduct (Matern and Fleming 1995) and the shimofuri and the shokihaze goby are found within the same range in the San Francisco Estuary and both have pelagic larvae susceptible to entrainment. One shokihaze goby was found on 22 August 2001 at the John E. Skinner Delta Fish Protective Facility (J. Morinaka, CDFG, personal communication), which is located at the northern end of the California Aqueduct. If the shokihaze goby finds its way to southern California, it may have ecological effects such as preying upon and disrupting feeding and spawning of the tidewater goby.

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[2]Wang, J. C. S. 1982. Early life history and protection of the tidewater goby, *Eucyclogobius newberryi* (Girard) in the Rodeo Lagoon of the Golden Gate National Recreation Area. Cooperative National Park Resources Studies Unit and University of California at Davis. Technical Report No. 7, 24 pp.

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DISTRIBUTION OF FEMALE NORTHERN PINTAILS IN RELATION TO HUNTING AND LOCATION OF HUNTED AND NON-HUNTED HABITATS IN THE GRASSLAND ECOLOGICAL AREA, CALIFORNIA

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To provide baseline information and better understand distribution of northern pintails (*Anas acuta*) in relation to hunting and distribution of hunted and non-hunted habitats in the Grassland Ecological Area (EA), we radio-tagged 191 Hatch-Year (HY) and 228 After-Hatch-Year (AHY) female pintails in the San Joaquin Valley and tracked their movements in the Grassland EA during September - March 1991-94. We investigated how the relative importance of public (National Wildlife Refuges [NWRs], California Department of Fish and Game Wildlife Areas [WAs]) vs. private areas (e.g., waterfowl hunting clubs [Clubs]), use of individual areas, and night destinations from specific day-use areas varied within and among winters and with pintail age and capture location. Overall, 64% of day and 85% of night pintail locations in the Grassland EA were on private areas. Day use of private areas was greater during nonhunting weeks (73% of all locations) and nonshoot days of hunting weeks (62% of all locations) than during shoot days (17% of all locations), when most pintails used public area sanctuaries. The effect of hunting lingered but faded, with use of private areas 1 day after hunting slightly less than 2 days after hunting (57% vs. 66%). Use of private areas on nights during nonhunting weeks and after nonshoot days of hunting weeks was identical (86%). Night use after shoot days was lower (79%), although the difference was significant only during the opening week of hunting. AHY females stayed on sanctuaries at a higher rate at dusk on shoot days and used the East Grassland (EGL) more than HY females. Use of private areas during the hunting season declined in 1993-94, possibly because newly-restored public area habitats attracted pintails. Night use of private areas during nonhunting weeks was lowest during 1991-92, the year drought prevented summer irrigations, and

probably reduced food production on most private but few public wetlands. San Luis NWR was the most important shoot day sanctuary but Kesterson NWR use increased after wetlands in its sanctuary were restored. Merced NWR was the only public area receiving high use at night during the hunting season. Most pintails that left Merced NWR at night flew to South Clubs rather than to closer East or North Clubs. Few pintails from Kesterson NWR flew to South Clubs at night but San Luis NWR and Los Banos WA pintails used both North and South Clubs at night. The percentage of San Luis NWR pintails going to South Clubs increased after the first hunting interval, except in 1993, when 10% instead went to newly restored Salt Slough WA watergrass (*Echinochloa crusgalli*) marsh. Pintails were more likely to use areas during hunting season that they frequented during August - October before hunting began, indicating that early habitat conditions influenced pintail use later in winter. Pintail distribution changed among intervals and years in response to changing hunting pressure and distribution of hunted and nonhunted habitats. Our data can serve as a baseline to evaluate response of pintails to changes in habitat management in the Grassland EA.

INTRODUCTION

The Grassland Ecological Area (EA) in the northern San Joaquin Valley (SJV) is the largest contiguous block of wetland habitat remaining in California's Central Valley and provides critical habitat for many wetland-dependent species (Grassland Water District 1999), including northern pintails (*Anas acuta*) (hereafter referred to as "pintails"). About half of the pintails in North America winter in the Grassland EA and other Central Valley habitats (Bellrose 1980, U. S. Fish and Wildlife Service[1] [USFWS] 1978), arriving as early as the first week of August and remaining through March. Pintail populations in North America declined to all time lows in the early 1990s (USFWS and Canadian Wildlife Service[2] [CWS] 1995) and abundance in California during winter was about 25% of that recorded in the 1970s (Pacific Flyway waterfowl reports and USFWS, Portland, Oregon, unpublished data).

Understanding pintail distribution and movements in the Grassland EA in relation to hunting pressure and location of hunted and nonhunted areas (i.e., sanctuaries) during winter is crucial to managing pintail populations and habitats. Grassland EA is a focal point for habitat conservation efforts (USFWS and CWS[3] 1986, Central Valley

[1]U.S. Fish and Wildlife Service. 1978. Concept plan for waterfowl wintering habitat preservation, Central Valley, California. U.S. Fish and Wildlife Service, Portland, Oregon, USA.

[2]U.S. Fish and Wildlife Service and Canadian Wildlife Service. 1995. Waterfowl population status, 1995. U.S. Fish and Wildlife Service, Washington, D.C., USA.

[3]U.S. Fish and Wildlife Service and Canadian Wildlife Service. 1986. North American waterfowl management plan - a strategy for cooperation. U.S. Fish and Wildlife Service, Washington, D.C., USA.

Habitat Joint Venture Implementation Board[4] 1990) and knowledge of pintail movements before habitat changes occur is necessary to evaluate habitat programs. Further, most Grassland EA wetlands are privately owned and managed with funds derived largely from hunters (Gilmer et al. 1982). Changes in pintail movements that impact hunter success could impact management of many Grassland EA wetlands (Heitmeyer et al. 1989, Baldassarre and Bolen 1994).

To identify factors related to pintail distribution and provide baseline information necessary for evaluating impacts of future changes in hunting and distribution of hunted and non-hunted habitats in the Grassland EA, we radio-tagged Hatch-Year (HY) and After-Hatch-Year (AHY) female pintails throughout the SJV after their late summer arrival and monitored their movements in the Grassland EA during late August to late March, 1991-94. Waterfowl surveys (California Department of Fish and Game [CDFG], Los Banos, California, unpublished data) provide some information on diurnal distribution but most were conducted on hunting days when pintails were concentrated on Wildlife Area (WA) and National Wildlife Refuge (NWR) sanctuaries. We studied distribution of individual pintails throughout the wintering period, during both day and night (when pintails primarily feed during most of the winter [Miller 1985, Euliss[5] 1984]), relative to hunting (shoot and nonshoot days) and location of hunted and nonhunted habitats.

STUDY AREA

The Grassland EA (Fig. 1) comprised the north (NGL), south (SGL) and east grasslands (EGL) and nearby San Luis Reservoir with forebay. The NGL were comprised of public areas with some wetlands closed to hunting (San Luis NWR [532 - 570 ha wetlands in sanctuary], Kesterson NWR [99 - 138 ha wetlands in sanctuary], Los Banos WA [196 - 219 ha wetlands in sanctuary), public areas without closed zones (Volta, Salt Slough, and China Island WAs) and privately owned waterfowl hunting clubs (North Clubs). The Grassland State Park in the NGL was closed to hunting but had no flooded areas. The SGL were entirely private (South Clubs). The EGL were composed of Merced (174 - 254 ha wetlands in sanctuary) and Arena Plains NWRs (49 ha wetlands, all sanctuary) and waterfowl hunting clubs (East Clubs). Overall during 1991-94, private area flooding comprised an average of 75% of all available habitat before hunting season (i.e., Prehunt) and 82% thereafter (Fig. 2).

Most wetlands were unflooded during the summer but irrigated periodically to promote seed production, then flooded during fall and winter. Initial flooding of most wetlands occurred during mid-August to late-October. Water for irrigation, fall flood-

[4]Central Valley Habitat Joint Venture Implementation Board. 1990. Central Valley Habitat Joint Venture Implementation Plan - a component of the North American Waterfowl Management Plan. U.S. Fish and Wildlife Service, Portland, Oregon, USA.

[5]Euliss, N.H., Jr. 1984. The feeding ecology of pintail and green-winged teal wintering on Kern National Wildlife Refuge. M.S. Thesis, Humboldt State Univ., Arcata. 188 pp.

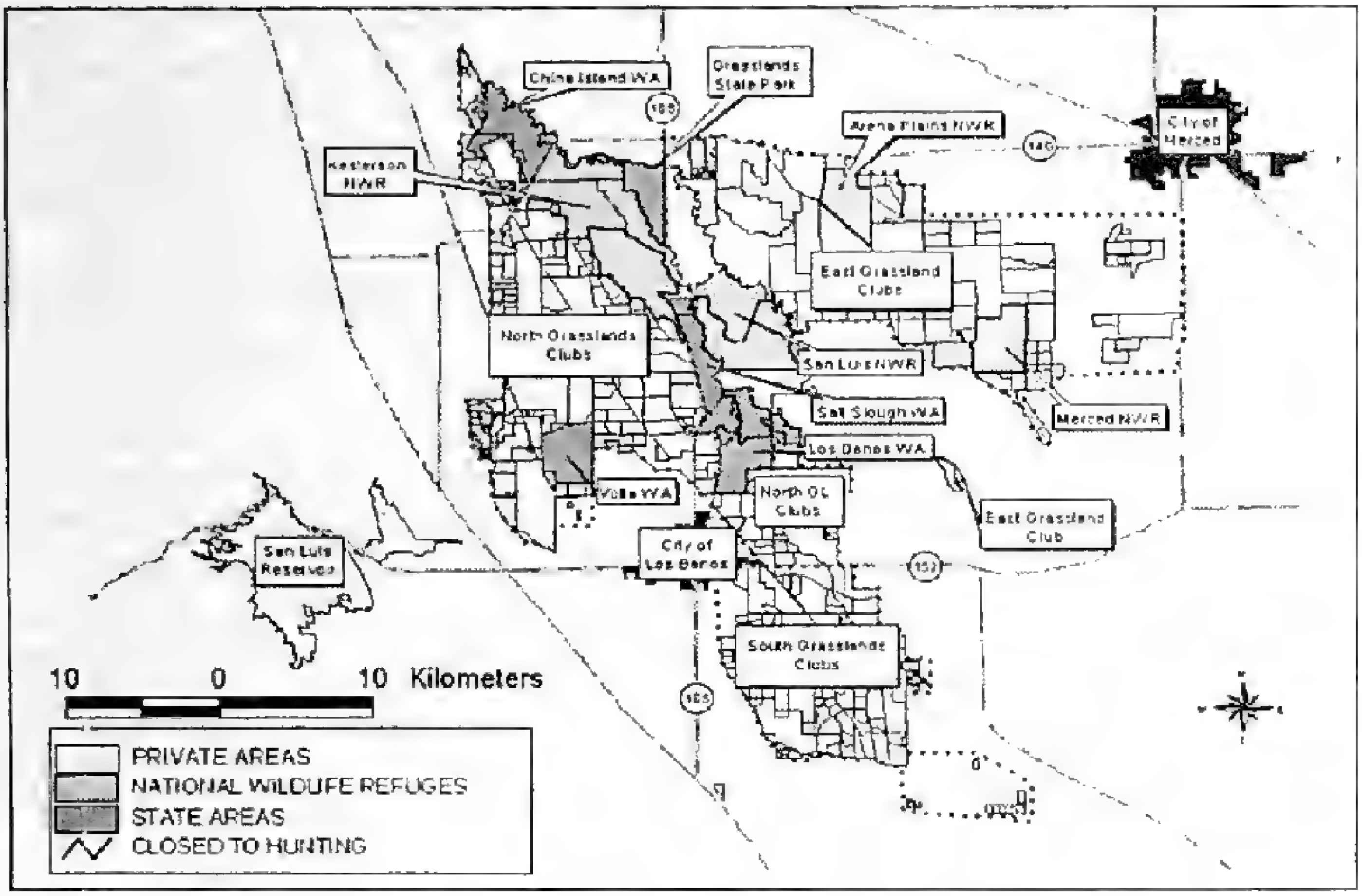

Figure 1. Grassland Ecological Area in the San Joaquin Valley, including California Department of Fish and Game Wildlife Areas (WAs), U.S. Fish and Wildlife Service National Wildlife Refuges (NWRs), private waterfowl hunting clubs, and San Luis Reservoir, during 1991-94.

up, and winter maintenance was delivered from reservoirs that stored Sierra Nevada snow-melt. Thus, the amount of early-winter habitat varied as a result of the previous winter's snowfall. Winter rains flooded additional habitat each year.

Changing precipitation, water availability, and management affected habitat availability. Drought conditions in the San Joaquin River drainage were the worst on record during 1991-92 (California Department of Water Resources[6] 1991, National Oceanic and Atmospheric Administration, Asheville, NC, unpublished data) and no water was available for May - July private wetland irrigation. Fall flood-up was delayed 2 weeks and record low August through mid-November water was delivered to private wetlands (Grassland Water District, Los Banos, CA, unpublished data). Conditions improved after January 1992 because of normal to above-average precipitation. Conditions during 1993-94 improved further because the Central Valley Project Improvement Act (Davis 1992) nearly doubled the amount of water delivered to the Grassland Water District (Grassland Water District, Los Banos, CA, unpublished data)

[6]California Department of Water Resources. 1991. California's continuing drought, 1987-1991: A summary of impacts and conditions as of December 1, 1991. State of California, Sacramento, California, USA.

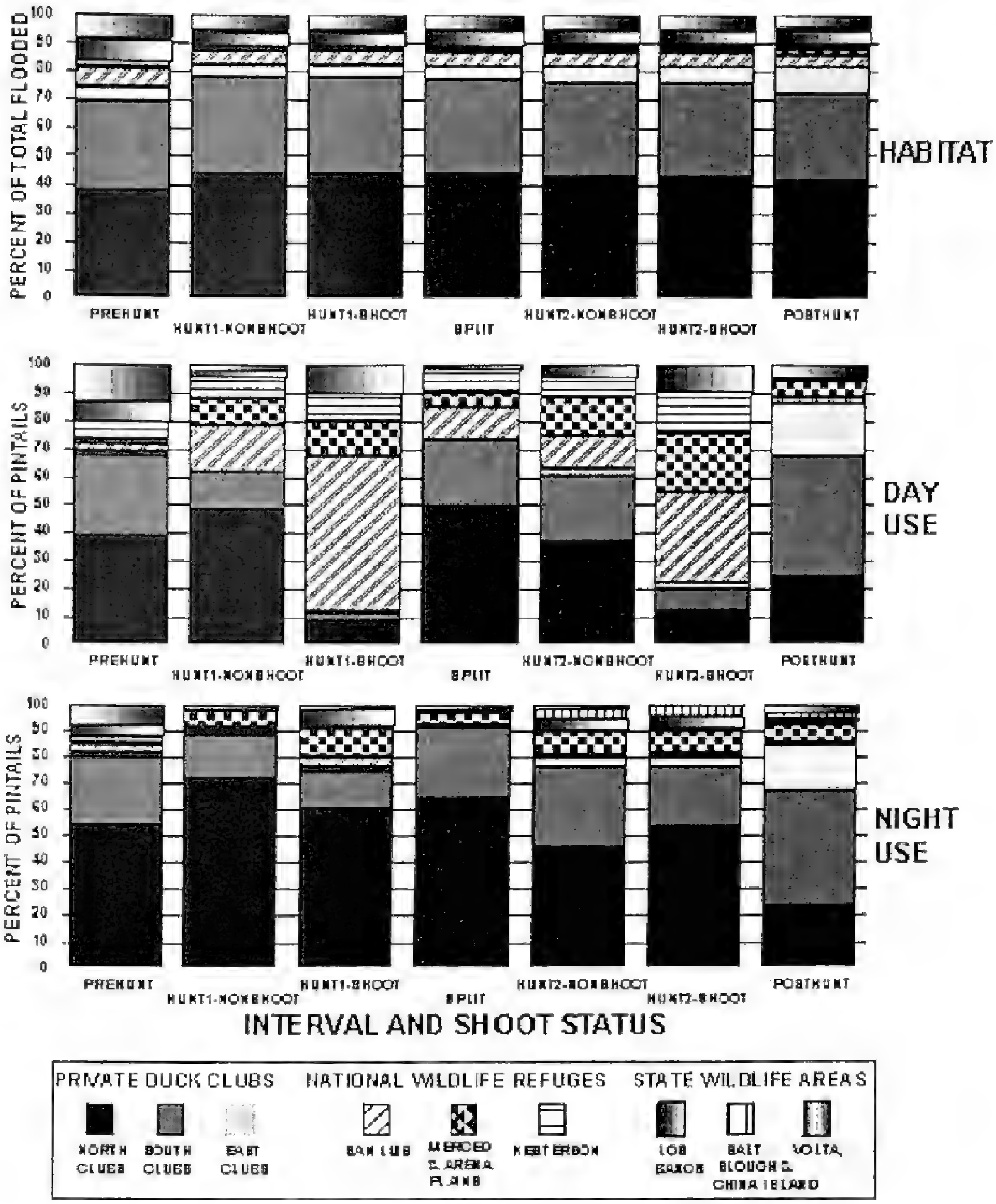

Figure 2. Percentage of total flooded habitat (excluding San Luis Reservoir and Forebay) and radio-tagged female northern pintail (*Anas acuta*) day and night locations on public areas (California Department of Fish and Game Wildlife Areas [WA], U.S. Fish and Wildlife Service National Wildlife Refuges [NWR]), and private waterfowl hunting clubs in the north, south, and east parts of the Grassland Ecological Area before (Prehunt), between (Split), after (Posthunt), and on shoot (Wednesdays and weekends) and nonshoot days during the first (Hunt1) and second (Hunt2) duck hunting seasons, 1991-94. Most pintail use was on Merced NWR but also includes use of Arena Plains NWR. Most use was on Salt Slough WA during Hunt2 but also includes use of China Island WA. San Luis Reservoir (not listed in legend) was used by only 1 or 2 birds for 3 Posthunt weeks in 1991-92 and 3 Hunt2 weeks in 1992-93.

and wetlands were restored on Salt Slough WA and the Kesterson NWR sanctuary. Mean weekly availability of marsh (89% to 96% seasonal marsh) during Prehunt was 6,088 ha in 1991-92, 7,431 ha in 1992-93 and 10,402 ha in 1993-94. These values increased to 20,106 ha, 20,815 ha and 23,723 ha respectively during the hunting season; and 20,863 ha, 22,318 ha and 24,473 ha respectively after the hunting season. Flooding of uplands and harvested fields before (46 ha, 105 ha, 207 ha) and during the hunting season (196 ha, 231 ha, 253 ha) followed similar annual patterns but availability after hunting was much greater in 1992-93 (1,178 ha) and 1993-94 (1,472 ha) than in 1991-92 (294 ha) because of variation in winter rains. Availability of reservoirs (6,348 - 6,372 ha), sewer ponds (194 - 245 ha) and evaporation ponds (1 - 39 ha) varied less among years and seasons. Habitats are described by USFWS[1] (1978), Heitmeyer et al. (1989), Johnson et al. (1993), and Baldassarre and Bolen (1994).

Daily bag limits (4 ducks with 1 either-sex pintail) and hunting season lengths (59 days) were constant but timing of seasons varied (CDFG, Sacramento, California, unpublished data). The hunting season (Hunt) was split with a 22-day first season (Hunt1) starting 26 (1991), 24 (1992) or 23 October (1993), and a 37-day second season (Hunt2) starting after a 12- (1991), 19- (1992) or 27-day (1993) closure (Split). Nearly all clubs, WAs and NWRs allowed hunting only on Wednesdays, Saturdays, and Sundays (i.e., shoot days). We define Posthunt as the interval from end of Hunt2 to 1 April 1992 and 1993 or 17 March 1994.

METHODS

Field Procedures

We captured and radio-tagged 228 AHY and 191 HY female pintails 29 August - 6 October 1991, 31 August - 5 October 1992 and 28 August - 25 September 1993 with 11 - 14 rocket-net (Schemnitz 1994) shots each year in the Grassland EA (Volta and Los Banos WAs; San Luis and Kesterson NWRs; South Clubs), Mendota WA (50 km southeast of SGL), and Tulare Lake Bed-Kern NWR vicinity of the Tulare Basin (100 km southeast of Mendota WA) (Table 1). We aged (HY or AHY, Larson and Taber 1980, Duncan 1985, Carney[7] 1992), legbanded, and released all pintails at the capture location <1 to 19 ($\bar{\chi}$ = 7.7) hours after capture. During the first 2 years we attached exclusively 20-21-g radio transmitters with back-mounted harnesses (Dwyer 1972). In 1993, we radio-tagged 98 pintails with the harness and 83 pintails with a spear-suture transmitter (similar to Pietz et al. [1995], except 8-9 g and 20 mm diameter x 12 mm high). Each transmitter had a unique signal, a mortality sensor, life expectancy ≥210 days and an initial minimum ground-to-ground range of 3.2 km using 150-db receivers and a dual 4-element Yagi null-peak telemetry system (Cochran and Lord 1963) mounted on the roof of pick-up trucks. Transmitters were imprinted with contact information which we

[7]Carney, S.M. 1992. Species, age and sex identification of ducks using wing plumage. U.S. Fish and Wildlife Service, Washington, D.C., USA.

Table 1. Number of After-Hatch-Year (AHY) and Hatch-Year (HY) female northern pintails (*Anas acuta*) radio-tagged in the Grassland Ecological Area (South Clubs, Volta Wildlife Area [WA], Los Banos WA, San Luis National Wildlife Refuge [NWR], Kesterson NWR), Mendota WA, and Tulare Basin of the San Joaquin Valley, California, 1991-93.

	Year and Age Class							
	1991		1992		1993		Total	
Area	AHY	HY	AHY	HY	AHY	HY	AHY	HY
South Clubs	17	13	19	15	19(8)	20(9)	55(8)	48(9)
Volta WA	8	5	10	26	0	1(0)	18(0)	32(0)
Los Banos WA	14	17	1	1	0	0	15(0)	18(0)
San Luis NWR	2	2	0	0	0	0	2(0)	2(0)
Kesterson NWR	0	0	0	6	25(12)	18(9)	25(12)	24(9)
Mendota WA	21	4	17	4	33(14)	39(19)	71(14)	47(19)
Tulare Basin	10	2	18	6	14(6)	12(5)	42(6)	20(5)
All Areas	72	43	65	58	91(40)	90(42)	228(40)	191(42)

( )Number of spear-suture type radio-tags in parenthesis, included in cell totals. All other radio-tags were harness backpack type.

solicited by posting project descriptions at hunting check stations and in state-wide media.

We scanned the entire study area, and for each pintail present, determined their location on ≥2 shoot days and nights and ≥2 nonshoot days and nights each week during Hunt and ≥2 days and nights each week during other intervals. We conducted aerial searches (Gilmer et al. 1981) weekly to ensure we found all pintails. In addition to the pintails we radio-tagged, we also tracked 25 AHY and 24 HY female pintails radio-tagged in Suisun Marsh (Casazza[8] 1995) and 3 AHY female pintails radio-tagged in Alaska (J. B. Grand, personal communication) while they were in the Grassland EA. We obtained 2 bearings from known locations to minimize time between bearings and because preliminary testing showed more bearings did not increase accuracy. The road network allowed us to obtain >90% of all locations <1.6 km from the bird at 50-130 degree angles. Warnock and Takekawa (1995) reported an average azimuth error of 1.5° and an error polygon of 1.1 ha with location distances 0.5 - 3.0 km using a system identical to ours. We entered truck location and azimuth, bird ID and azimuth, time, date, observer and truck ID, and calculated bird locations using a modified version of XYLOG and UTMTEL (Dodge et al.[9] 1986, Dodge and Steiner 1986). We intersected pintail locations in a Geographic Information System with digitized maps and identified the polygon (average size = 20.3 ha) associated with each location.

[8]Casazza, M.L. 1995. Habitat use and movements of northern pintails wintering in the Suisun Marsh, California. Thesis, California State University, Sacramento, California, USA.

[9]Dodge, W.E., D.S. Wilkie, and A.J. Steiner. 1986. UTMTEL: A laptop computer program for location of telemetry "finds" using Loran-C. Massachusetts Cooperative Research Unit. Report, U.S. Fish and Wildlife Service.

Data Analysis

We summarized pintail use in the Grassland EA for private (Clubs) vs. public areas (NWRs and WAs), individual areas (e.g., Kesterson NWR, North Clubs), and report on night destinations of pintails from specific day-use areas. We used categorical modeling (Sauer and Williams 1989) to investigate the relationship between weekly pintail distribution and diurnal period (day vs. night), study year (1991-92 vs. 1992-93 vs. 1993-94), bird age (HY vs. AHY), and bird capture location (Grassland EA vs. other [Mendota WA, Tulare Basin, Suisun Marsh, Alaska]; or NGL vs. SGL). We conducted most analyses separately for hunting and non-hunting weeks. During hunting weeks, we compared distribution on shoot vs. nonshoot days, days or nights following a shoot day vs. two days after a shoot day, and week days. We used PROC GENMOD (SAS Institute 1997) that conducts a Chi-square test for overall differences and a Z test for individual differences across weeks with a generalized estimating equations approach (McCullagh and Nelder 1989) to account for correlation between repeated measures (Liang and Zeger 1986). We used PROC CATMOD (SAS Institute 1989) by week, with a Chi-square test and the Bonferroni adjustment to maintain an alpha level of 0.05 (Johnson and Wichern 1982), to test for consistency among weeks for variables that were found to be significant across weeks. We followed Dobson (1990) and Milliken (1984) to assess the importance of explanatory variables and interactions using a step-down model selection method. To reduce bias associated with unequal and multiple sampling of individual pintails each week, we apportioned multiple day, night, shoot and nonshoot locations among areas so that each pintail had a maximum of one location per week for each day, night, shoot and nonshoot category. For instance, if during week 9 in Hunt1, a pintail was located on San Luis NWR during the day on Wednesday and on Merced NWR during the day on Saturday, we apportioned 0.5 shoot day locations to each of those areas. We grouped weekly totals into intervals (Prehunt, Hunt1, Split, Hunt2, Posthunt) and intervals into hunting (Hunt1 and Hunt2) and nonhunting (Prehunt, Split, Posthunt) for some analyses. To pool or compare weekly distribution across years, we used 1 September, 30 August, or 29 August as the start of week 1 for 1991-92, 1992-93 and 1993-94, respectively. We conducted a nearest neighbor analysis (Rosing et al. 1998) and verified that each pintail moved about independently even if captured under the same net (Fleskes[10] 1999).

We report actual use proportions for most comparisons except we report difference in relative use (with 95% CI) for the few instances (i.e., impact of capture location, age class, week day) when the magnitude of difference is more meaningful than actual use proportions. For instance, difference in relative use of private areas on Wednesdays vs. Saturdays was calculated as: (proportion of Wednesday use occurring on private areas/ 1 - proportion of Wednesday use occurring on private areas) / (proportion of Saturday use occurring on private areas/ 1 - proportion of Saturday use occurring on

[10]Fleskes, J.P. 1999. Ecology of female northern pintails during winter in the San Joaquin Valley, California. Dissertation, Oregon State University, Corvallis, Oregon, USA.

private areas). The difference in relative use is a less biased measure of the impact of a variable than the difference in actual use proportions because relative use can range from 0 to infinity so that an equal percentage difference in relative use has the same meaning regardless of the values of relative uses being compared. In contrast, proportions are restricted to between 0 and 1, so that the meaning of a percentage change in proportions depends on the values of the proportions being compared (e.g., a 50% increase in 10% differs in meaning from a 50% increase in 20%).

RESULTS

Use of Private vs. Public Areas

Overall during September through March, 64% of day and 85% of night locations in the Grassland EA were on private areas (i.e., North, South and East Clubs). However, the relative importance of public and private areas and factors related to use patterns varied greatly among intervals with and without hunting (Fig. 2).

Nonhunting Weeks

During weeks of nonhunting intervals (Prehunt, Split, Posthunt), the relative importance of public and private areas differed between day and night ($X^2 = 200.02$, 6 df, $P < 0.0001$) and among study years ($X^2 = 30.33$, 10 df, $P = 0.0007$). Averaged across all nonhunting weeks and years, use of private areas during day was slightly less than at night (73% vs. 86% of locations; $Z \geq 4.31$, $P < 0.0001$). However, the strength of the diurnal effect varied among weeks, so that day and night use differed significantly during 3 of 5 Prehunt ($X^2 \geq 4.65$, 2 df, $P \leq 0.03$), all (3/3) Split ($X^2 \geq 13.65$, 1 df, $P < 0.001$), but no (0/5) Posthunt weeks ($X^2 \leq 0.11$, 1 df, $P \geq 0.74$) that we tested (samples too small to test other weeks). The relative importance of public and private areas during day was similar all years ($Z < 1.18$, $P > 0.24$) but night use of private areas during nonhunting weeks in 1991-92 (76.2%) was less than in 1992-93 (87.2%; $Z = 2.12$, $P = 0.034$) and 1993-94 (89.3%; $Z = 3.30$, $P = 0.0009$). The year effect was significant during 3 of 5 Prehunt, 3 of 5 Posthunt ($X^2 \geq 6.26$, 2 df, $P \leq 0.04$) but no Split weeks ($X^2 \leq 0.79$, 2 df, $P \geq 0.67$).

Hunting Weeks

The relative importance of public and private areas changed drastically once hunting began (Fig. 3), with most pintails seeking sanctuary in parts of public areas closed to hunting and flying out at dusk to private clubs (Fig. 4). Proportions of pintails on public and private areas during hunting weeks were related to diurnal period ($X^2 = 859.58$, 7 df, $P < 0.0001$), hunt status (i.e., shoot vs. nonshoot; $X^2 = 1011.88$, 7 df, $P < 0.0001$), weekday ($X^2 = 42.81$, 14 df, $P < 0.0001$), study year ($X^2 = 24.81$, 12 df, $P < 0.0157$) and bird age ($X^2 = 26.99$, 7 df, $P = 0.0003$).

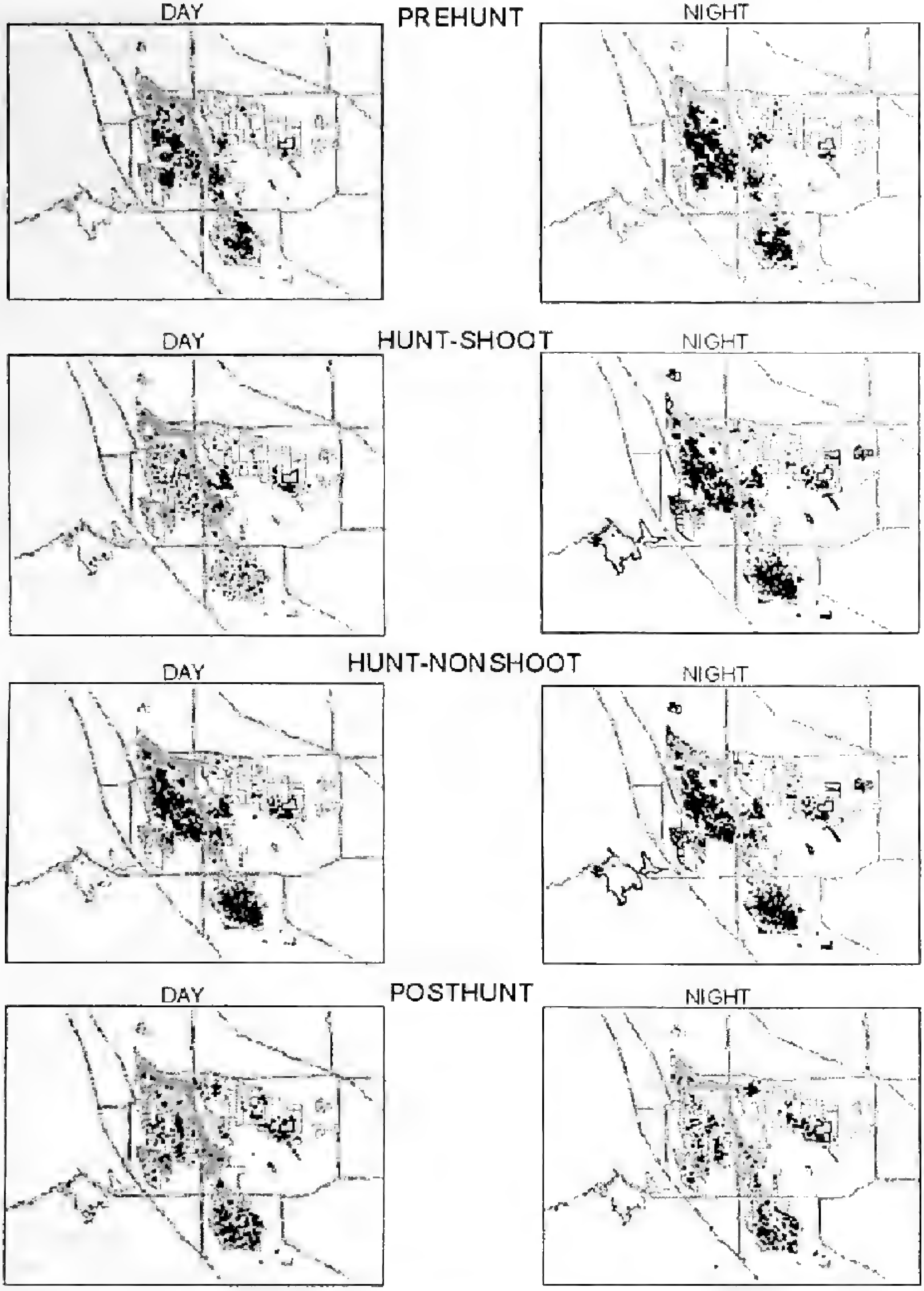

Figure 3. Day and night locations (dots) of radio-tagged female northern pintails (*Anas acuta*) in the Grassland Ecological Area before (Prehunt), after (Posthunt), and on shoot (Wednesdays and weekends) and nonshoot days during duck hunting season (Hunt), 1991-94. Locations during the Split between the first and second hunting intervals included in Hunt-Nonshoot.

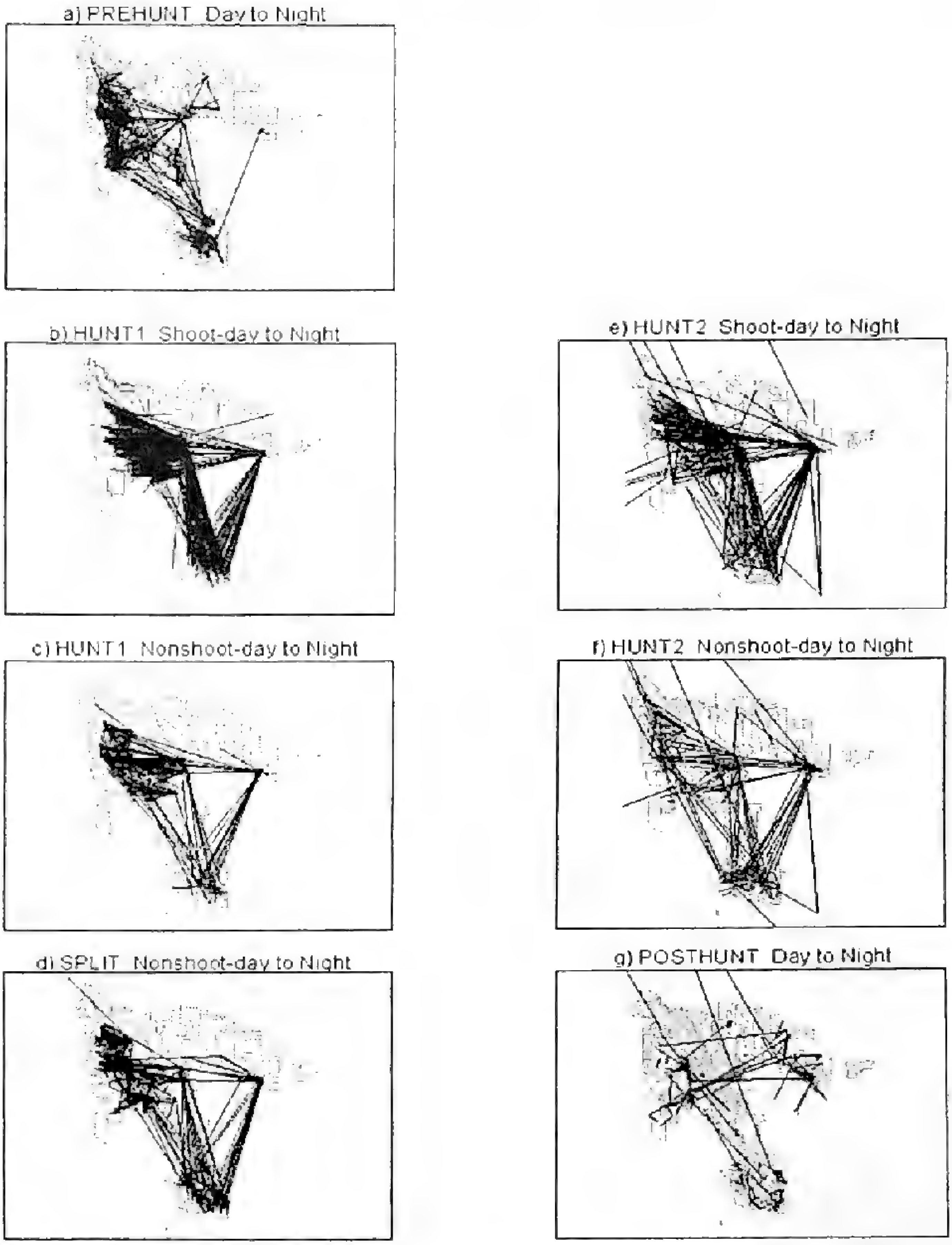

Figure 4. Movements of radio-tagged female northern pintails (*Anas acuta*) in the Grassland Ecological Area between day and night locations during Prehunt, Split, Posthunt, and on shoot (Wednesdays and weekends) and nonshoot days during Hunt1 and Hunt2, 1991-94.

Overall during hunting intervals, pintail use of private areas during day was about half that at night (42% vs. 83%; $Z = 14.83$, $P < 0.0001$). The diurnal difference was much greater than during nonhunting intervals because the difference in use between shoot and nonshoot days (17% vs. 62%; $Z = 30.35$, $P < 0.0001$) was much greater than between nights following a shoot day and nights after nonshoot days (79% vs. 86%; $Z = 4.33$, $P < 0.0001$). The difference in private use on shoot and nonshoot days was greater during Hunt1 weeks ($X^2 \geq 95.93$, 1 df, $P < 0.0001$) than Hunt2 weeks ($X^2 \geq 4.75$, 1 df, $P \leq 0.03$); at night differences between shoot and nonshoot were significant ($X^2 \geq 17.75$, 1 df, $P \leq 0.0001$) only during the first week of Hunt1. Shoot day use of private lands was similarly low ($Z = 0.75$, $P = 0.45$) for AHY and HY pintails but relative use of private areas on nights after a shoot day was 43% (16% to 61%) lower for AHY pintails than HY pintails.

The impact of hunting lingered into nonshoot days but faded over time, with use of private areas 1 day after a shoot day less than 2 days after a shoot day (57% vs. 66%; $Z = 6.22$, $P < 0.001$). The lower use of private areas 1 vs. 2 days after hunting was significant during all Hunt1 weeks ($X^2 \geq 4.61$, 1 df, $P \leq 0.03$), but no Hunt2 weeks ($X^2 \leq 1.99$, 1 df, $P \geq 0.16$). The use of private areas on the second and third night after a shoot day were similar.

Use patterns differed for some weekdays. Among nonshoot days, proportions of pintail day use on private and public areas on Mondays was similar ($X^2 = 8.74$, 7df, $P = 0.27$) to Thursdays (both 1 day after shooting) but Tuesday and Friday distribution (both 2 days after shooting) differed some years ($X^2 = 53.70$, 7df, $P < 0.0001$). Distribution on Tuesdays and Fridays was similar in 1991-92 ($Z = 0.84$, $P = 0.40$), but in 1992-93 the relative use of private areas on Friday was 53% (42% to 63%) lower than on Tuesday; in 1993-94 the relative use of private areas on Friday was 89% (39% to 157%) greater than on Tuesday. Among shoot days, the relative use of private areas on Wednesdays was 82% (35% to 146%) greater than on Saturdays and 62% (23% to 112%) greater than on Sundays. Relative use of private areas during the day on Saturdays and Sundays were similar ($Z = 0.77$, $P = 0.44$). Relative use of private areas on Wednesday night was 66% (23% to 124%) greater than on Saturday nights and 194% (89% to 622%) greater than on Sunday night. Relative use of private areas on Saturday nights was 37% (12% to 301%) greater than on Sunday nights.

Shooting reduced use of private areas all 3 years ($Z \geq 12.10$, $P < 0.0001$) but there was weak evidence that private area use was less in 1993-94 than earlier years. Averaged across day-night, use of private areas on shoot dates in 1993-94 (45.6%) tended to be lower than in 1992-93 (51.3%; $Z = 2.97$, $P = 0.0029$) and 1991-92 (48.7%; $Z = 1.80$, $P = 0.07$); on nonshoot dates the relative importance of private areas was similar each year ($Z \leq 1.41$, $P \geq 0.16$).

Use of Specific Areas

Pintail use of specific private (North Clubs, South Clubs, East Clubs) and public (Merced-Arena Plains NWRs, San Luis NWR, Kesterson NWR, Los Banos WA, Volta-Salt Slough-China Island WAs) areas in the Grassland EA varied among intervals (Fig. 2).

Prehunt

During Prehunt, 68% of day use and 80% of night use occurred on North and South Clubs (Fig. 2 and 3). The percentage of pintails using South Clubs was similar ($t = 1.68$, Bonferroni $P > 0.05$) during the day and night but use of the North Clubs was greater ($t = 7.35$, Bonferroni $P < 0.05$) at night than during the day (Fig. 2). Most of the additional pintails in North Clubs at night came from Kesterson NWR, Volta WA and Los Banos WA, where day use was roughly double ($t \geq 2.80$, Bonferroni $P < 0.05$) night use (Fig. 2). Use of other Grassland EA areas was low and similar ($t \leq 1.68$, Bonferroni $P > 0.05$) during day and night. The location within the Grassland EA where pintails were captured (i.e., NGL or SGL) was the most important factor related to their distribution during Prehunt. Relative use of the SGL was 48,900% (7,700% - 305,700%) greater for pintails radio-tagged in the SGL than pintails radio-tagged in the NGL. Likewise, relative use of the NGL was 11,600% (4,400% - 30,400%) greater for pintails radio-tagged in the NGL than pintails radio-tagged in the SGL. Thus, use of some areas varied greatly among years, reflecting differences in flooding and where we captured pintails. Relative use of the EGL during Prehunt was similar for the few pintails from NGL and SGL. Distribution within the Grassland EA during Prehunt was similar for pintails captured inside and outside (e.g., Mendota WA, Tulare Basin) the Grassland EA.

Hunt1

Pintail distribution changed drastically once hunting began (Fig. 3), with most pintails roosting on public area sanctuaries on shoot days and flying out at dusk to Clubs. San Luis NWR was the most common shoot day sanctuary (Fig. 2). Shoot day use in 1993-94 vs. earlier years was greater in San Luis NWR (61% vs. 52 to 55%) and Kesterson NWR (18% vs. 5 to 8%) but lower in Los Banos WA (6% vs. 11 to 12%) and Merced NWR (9% vs. 13 to 14%). Night and nonshoot day use shifted from areas farthest from sanctuaries (i.e., South Clubs and Volta WA) to areas closer to sanctuaries (i.e., North Clubs) (Fig. 2). North Clubs were the most common nonshoot day and night location for pintails (Fig. 2) but a greater percentage of pintails flew to the distant South Clubs in 1991-92 (21%) then later years (13 to 14%). Overall during Hunt, the relative use of the SGL was 104% (24% to 233%) greater for pintails radio-tagged in the SGL than NGL, indicating that pintails that used the SGL during Prehunt flew to the SGL from sanctuaries in the NGL and EGL more often than pintails that had not used the SGL during Prehunt. The relative use of the NGL on nonshoot nights was 103% (15% to 259%) greater for pintails radio-tagged in the NGL than those from the SGL, but because many SGL pintails remained in the NGL during Hunt, the trend was not significant on shoot days (7%, -41% to 95%), nonshoot days (58%, -9% to 175%) or shoot nights (37%, -24% to 147%). The relative use of the SGL was 51% (4% to 119%) greater for pintails radio-tagged outside the Grassland EA (e.g. Mendota WA, Tulare Basin) than for pintails captured within the Grassland EA. Use of NGL and EGL was similar for pintails captured in and outside the Grassland EA. Relative use of EGL was 60% (37% to 75%) greater for AHY than HY; relative use for AHY vs. HY did not differ significantly in NGL (39%, -2% to 97%) or SGL (-39%, -107% to 7%).

Split

During the 2-4 week Split, pintails responded to the lack of hunters by remaining on private clubs during day and night (Fig. 2). Use of South Clubs increased to near Prehunt levels but use of Volta and Los Banos WAs remained low. Day use of Kesterson and Merced NWRs was similar to Hunt1 but most birds that had been roosting on San Luis NWR did not return each morning as they did during Hunt1 and use there was lower (Fig. 2). Day use of Kesterson NWR during Split continued the trend observed during Hunt1, increasing over the 3 years from 7% to 9% to 11%. Both day (29%) and night (35%) use of the SGL in 1991-92 was greater than day (19 to 22%) and night (23 to 26%) use in later years.

Hunt2

Use patterns during Hunt2 were similar to Hunt1 except a greater percentage of shoot day locations were on private areas and importance of South Clubs, East Clubs and Merced NWR increased whereas importance of San Luis NWR and North Clubs decreased (Fig. 2). Annual trends in day use were similar to Hunt1 except shoot-day use of North Clubs during Hunt2 increased (10% to 12% to 15%) rather than decreased over the 3 years and nonshoot day use of Kesterson NWR did not increase over the 3 years. Likewise, annual trends in night use during Hunt2 were similar to Hunt1 except night use of Salt Slough WA increased greatly from <3% in 1991-92 and 1992-93 to 11% in 1993-94 and use of East Clubs declined during both day (5% to 3% to 1%) and night (10% to 5% to 1%) during the 3 years.

Posthunt

During Posthunt, the importance of South and East Clubs peaked as pintails abandoned most public areas with sanctuaries, except Merced NWR, and settled into private wetlands (Fig. 2). Although pintails used many of the same wetlands used during the hunting season, many habitats on the fringe of the Grassland EA received their first heavy use during Posthunt (Fig. 3). Most used the same area during the day and night (Fig. 3 and 4). Posthunt use over the 3 years increased on South Clubs (32% to 42% to 54%) but declined on East Clubs (32% to 18% to 5%). Use of North Clubs was lower in 1991-92 than later years (12% vs. 27 to 31%); day use of Volta WA was higher in 1991-92 than later years (8% vs. 3 to 4%).

Association Between Day- and Night-Use Areas

Night destinations of pintails in the Grassland EA differed among day-use areas, intervals, and years (Fig. 5). For private areas, 96% of the pintails using North and South Clubs and 83% using East Clubs during nonhunt and nonshoot days stayed there at night; 70% of the few that were on private areas on shoot days stayed there at night.

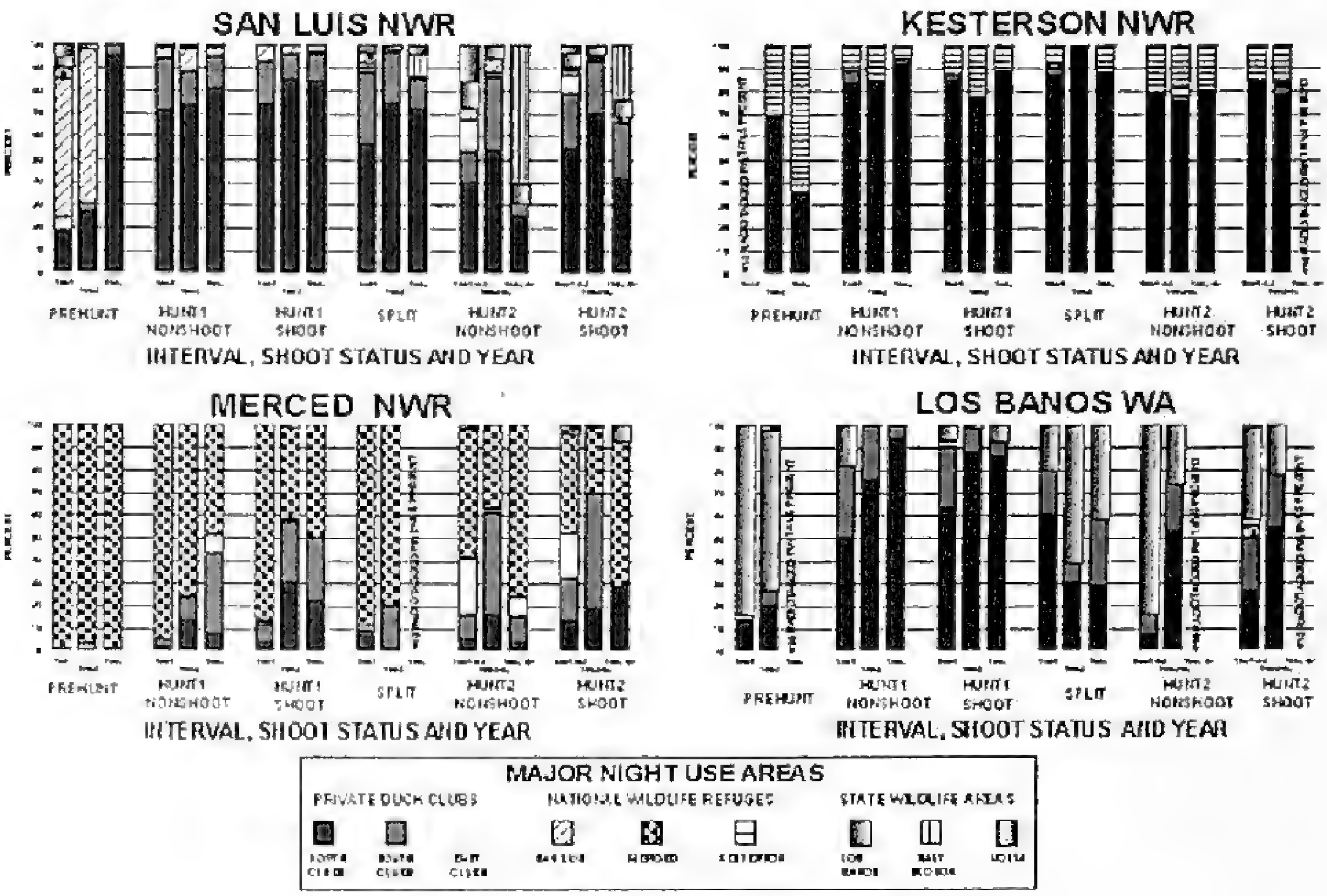

Figure 5. Percent of female radio-tagged northern pintails from 4 major day-use areas that were located at night in each area in the Grassland Ecological Area, before (Prehunt), between (Split), after (Posthunt), and on shoot (Wednesdays and weekends) and nonshoot days during the first (Hunt1) and second (Hunt2) duck hunting seasons, 1991-92, 1992-93, and 1993-94. Volta WA was a major day-use area only during Prehunt, when 69%, 57%, and 94% flew to North Clubs and 30%, 42%, and 3% stayed on Volta WA at night, during 1991, 1992, and 1993, respectively. Few pintails used private areas on shoot days; on nonhunt and nonshoot days, 96% of the pintails using North and South Clubs and 83% using East Clubs stayed there at night.

Except at Merced NWR, most pintails on public areas flew out to private areas at night (Fig. 5). Overall, a higher percentage of the Merced NWR pintails that left at night flew to South Clubs rather than to closer East or North Clubs; the percentage that left Merced for South and North Clubs during Hunt1 increased during the study (Fig. 5). Except during Prehunt, $\leq$10% of the pintails using San Luis NWR during the day stayed there at night. During Hunt1, >71% of the pintails on San Luis NWR went to North Clubs at night and 11-23% went to South Clubs. Thereafter, the percentage from San Luis NWR going to North Clubs decreased as the percentage going to South Clubs and, in 1993-94, to Salt Slough WA increased (Fig. 5). Few Kesterson NWR pintails went to South Clubs. During Prehunt, 31-65% of pintails using Kesterson NWR during day remained there at night and 35-69% flew to North Clubs; thereafter 67-99% flew to North Clubs at night. Night destinations of Los Banos WA pintails varied among intervals and the percentage during Hunt1 going to South Clubs decreased and to North Clubs increased over the 3-year study (Fig. 5).

DISCUSSION

Hunting and habitat distribution

Hunting and distribution of sanctuaries and other habitats were the main factors affecting pintail distribution during September - March in the Grassland EA.

During Prehunt, pintail distribution tracked distribution of flooded habitat. Variation in water supplies, location of canal repairs and other factors that impacted flooding schedules accounted for most variation in pintail distribution. Pintails that used the SGL during Prehunt were more likely to return there during the hunting season, indicating that Prehunt habitat distribution also influenced pintail distribution during the hunting season.

Variation in hunting pressure and distribution of both hunted and nonhunted habitats were important factors affecting pintail distribution during the hunting season. On shoot days, pintails were concentrated on public- area sanctuaries. Newly restored wetlands in the Kesterson NWR sanctuary attracted pintails and likely contributed to the decline in shoot-day use of private areas in 1993-94. At dusk, most pintails, except at Merced NWR, dispersed to private duck club wetlands where they remained until near dawn. This indicates that, by the start of hunting, the availability of preferred pintail food resources was lower on most public areas than on private clubs. With no sanctuaries nearby, pintail use of the SGL remained low until late winter, probably at which time food in habitats nearer sanctuaries became depleted and pintails were forced to make longer feeding flights. Restored Salt Slough WA wetlands adjacent to San Luis NWR, the main shoot-day sanctuary, likely contributed to the decline in night use of private lands in 1993-94. Most pintails remained on private areas on nonshoot days but a greater percentage returned near dawn to public sanctuaries on the day after hunting than 2 days after hunting. The higher return rate to sanctuaries the day after hunting was probably because some pintails simply continued their successful survival strategy of returning to refuges near dawn. However, we did not measure disturbance rates and disturbance due to hunters lingering on duck clubs or other factors may have been greater the day after hunting than 2 days after hunting. Other subtle differences in hunting pressure affected pintail distribution. Pintail use of private areas was greater on Wednesdays than weekends probably because most hunters were from outside the Grassland EA vicinity and fewer made the trip for a 1-day Wednesday hunt than a 2-day weekend hunt (Grassland Water District, Los Banos, California, unpublished data). Greater hunter success (CDFG and Grassland Water District, Los Banos, California, unpublished data) probably allowed hunters to bag limits and vacate wetlands earlier during Hunt2 than Hunt1, contributing to the increased pintail use of private areas that we observed during Hunt2. Fog that occurred mostly during Hunt2 (National Oceanic and Atmospheric Administration, Asheville, North Carolina, unpublished data) may have hidden pintails from hunters and also helped pintails remain on private clubs. Late-winter flooding and pintail use of EGL habitats varied with late-winter precipitation. Many habitats on the fringe of the Grassland EA received their first heavy use during Posthunt, indicating preferred foods were still available in those

habitats but had been depleted in most habitats nearer sanctuaries.

Habitat quality

In addition to habitat distribution, there was some evidence that habitat quality affected pintail distribution. Pintail use of private areas at night, when pintails primarily feed, was lowest during 1991-92. The continuing drought that year prevented summer irrigation of many private wetlands and probably reduced seed production in private wetlands below that of irrigated public wetlands. Waterfowl refuging theory predicts that when food supplies are low, feeding flight distances from sanctuaries will increase. Thus, low productivity of private wetlands in 1991-92 may also explain why a greater percentage of pintails flew to the distant South Clubs during Hunt1 and Split in that year compared to later years.

Pintail age

Age-related differences in individual experience may also have influenced pintail movements and distribution. AHY females left sanctuaries at a lower rate than HY females at dusk on shoot days, perhaps because AHY pintails were more likely to have experienced when hunters were still afield near dusk on shoot days. AHY females were also more likely than HY pintails to use the EGL, perhaps because most EGL habitats did not flood until late winter and HY females had no experience from prior years that those habitats became available with late-winter rains.

Limitations of our data

We used methods that minimized biases and allowed wide application of results. However, our findings should be considered in the light of our sampling methodology and other constraints. Logistics prevented us from studying males and applying our results to males should be done with an understanding that male and female movements may differ. We radio-tagged pintails throughout the SJV in accordance with surveyed abundance, but because it would have been difficult and disruptive to capture pintails throughout winter, we restricted trapping to Prehunt. Movements of pintails that arrive in the SJV after we trapped may differ from birds that we tracked. Our estimate of Prehunt distribution may have been biased because of our inability to capture pintails in all areas where they were abundant (i.e., Merced NWR, North Clubs; CDFG, Sacramento, California, unpublished data). However, the bias was probably not severe because radio-tagged pintails did move into the few pintail concentration areas that we did not sample. The low use of duck clubs measured on shoot days probably underestimated actual exposure of pintails to hunters. Pintails need only fly over or visit a hunted area briefly in order to risk being harvested whereas radio-tagged pintails must land in one location for several minutes for us to triangulate their location. Although missing a few minutes of use does not critically bias estimates of distribution, it does explain why

hunting mortality was high (Fleskes[10] 1999) relative to the low level of private-area use measured on shoot days.

MANAGEMENT IMPLICATIONS

Pintail distribution during this study changed primarily in response to changes in hunting pressure but also to distribution of hunted and nonhunted habitats. Our data can serve as a baseline to evaluate impacts to pintail distribution of future management changes in the Grassland EA. NWRs and WAs established during the last 30 years in the NGL and EGL have replaced San Luis Reservoir (CDFG, Sacramento, California, unpublished data) as the major pintail shoot-day roost site. Creating new sanctuaries in the future will also likely redistribute pintails. If new sanctuaries spread pintails more widely, then risk of catastrophic disease loss could be reduced. However, if pintail distribution or movements are changed in a way that reduces pintail harvest opportunity on some clubs, management of their habitats may change, impacting pintails and other wildlife dependent upon similar habitats. Understanding pintail movement can provide insight on likely impacts of habitat changes. For instance, knowing that many Merced NWR pintails fly to the SGL at night, we predict that habitat improvements in the EGL may reduce pintail abundance in the SGL. Establishing a SGL sanctuary would likely increase pintail use there.

To maximize pintail use of their areas, managers must provide attractive feeding and roosting habitats throughout the wintering period. Pintails were more likely to use areas during the hunting season that they frequented during Prehunt, indicating that early season habitat conditions influences pintail use later in winter. Opportunity to harvest pintails often arose during this study from pintails either staying in an area in the morning after feeding there at night or returning there in early evening to feed. Thus, the availability of adequate water supplies for summer irrigation to enhance production of waterfowl foods is crucial for a successful pintail hunting program. Most pintails left private clubs to roost elsewhere, even during nonhunting intervals and on nonshoot days, indicating that availability of diurnal roost sites may be limiting pintail use of private areas. Thus, providing additional undisturbed roosting sites on duck clubs would likely improve pintail use and harvest opportunity on duck clubs while also distributing pintails more widely throughout the Grassland EA and reducing risk of catastrophic disease losses.

An expanded program of nonshoot days would increase pintail use of private areas during the hunting season but may require enforced mandates. More pintails used private clubs on nonshoot than shoot days and on the second than first nonshoot day, indicating that an expanded nonshoot day program would increase pintail use of private areas. However, despite increased effort during 1992-93 to encourage voluntary compliance of nonshoot days, some clubs continued to hunt on some nonshoot days (Grassland Water District and Grasslands Resource Conservation District, unpublished data), and pintail use of private areas on nonshoot days did not increase. Thus, like other "tragedy of the commons" (Hardin 1968) situations, because the reward (i.e., duck harvest) for hunting on nonshoot days increases as duck abundance on private areas

increases, complete compliance will probably only be achieved when the penalties of hunting on nonshoot days exceed the rewards.

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